The Effects of Suspended Sediments on the Macroinvertebrate Community Structure of a River Ecosystem

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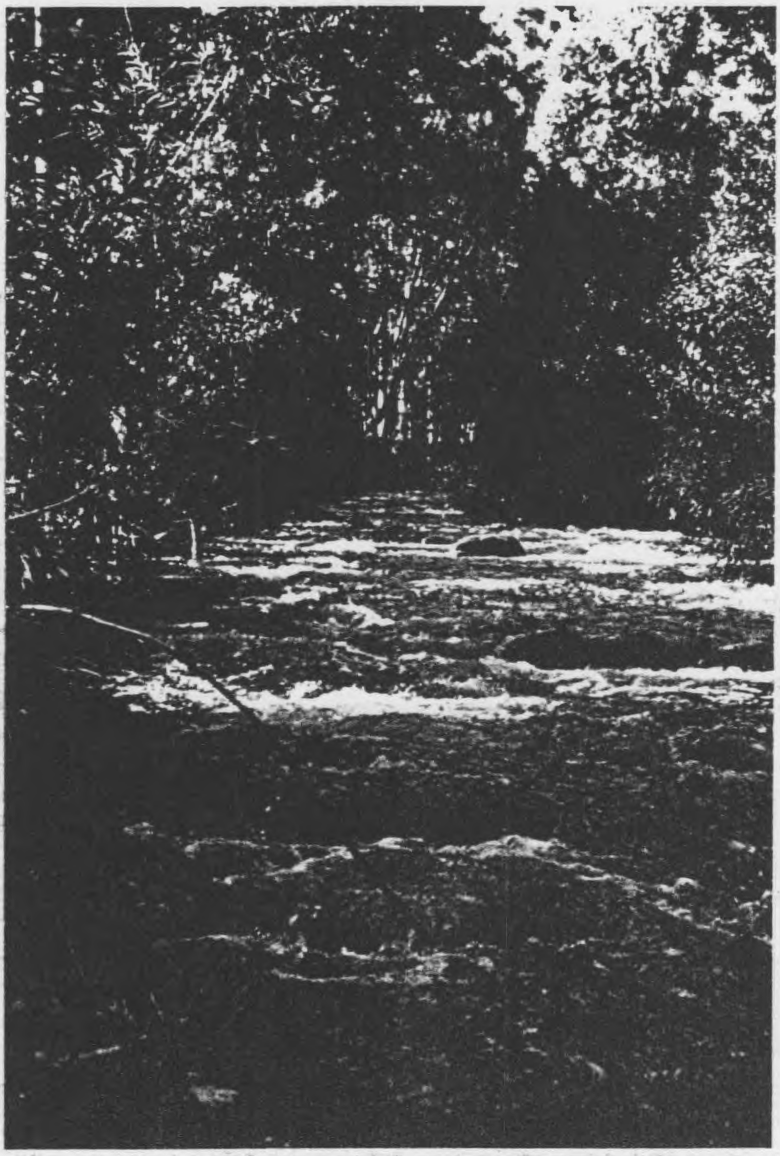


Plate 1. A view of the upper reaches Lourens River.

ABSTRACT

The faunal zonation patterns of the Lourens River, in the south western Cape, were established in late summer 1991, by sampling the macroinvertebrates down the length of the river, from the mountain reaches to the coastal plain. In winter, macroinvertebrate samples were collected from the upper river zone, above and below the source of high loads of suspended solids. These resulted from agricultural and land drainage practises in the adjacent area.

Classification procedures (cluster analysis and ordination by multi-dimensional scaling) were used to search for changes in community structure, and to compare these with physical and chemical variables.

The only abiotic change over the winter study reach was an increase in the concentration of suspended solids. Higher suspended solids were recorded below the silt sources, with up to a five-fold increases occurring after rainfall and spate activity. These changes were accompanied by the loss or drastic reduction in macroinvertebrate species characteristic of the mountain stream and upper river zones. The loss of Ephemeroptera species was particularly noticeable.

It was concluded that the changes in community structure were indicative of pollution stress by mainly inorganic solids in suspension. Some management options are suggested.

INTRODUCTION

Over the last decade there has been increasing recognition that the management of rivers necessitates the protection of their dynamic nature — the ability to change, and therefore also to recover from perturbation or forced change that results from pollution (HELLAWELL, 1977). This has been accompanied by a shift from chemical monitoring alone to developing biological monitoring programmes which assess the impacts of perturbations on the affected ecosystems (HAWKES, 1982; ARMITAGE *et al.*, 1991).

Whilst a chemical analysis of water quality provides useful information as to the nature of effluents entering the system, physical and chemical surveillance is generally discontinuous, providing only an instantaneous reflection of possible pollutants to an ecosystem. A biological approach examines organisms whose exposure to the water (and pollutants) is continuous. This approach can be used to quantify the structure and function of the ecosystem (HERRICKS *et al.*, 1981), and thus to detect environmental changes. This is important in pollution monitoring, where unknown, intermittent or unmeasured pollutants may go undetected by periodic chemical sampling.

The use of macroinvertebrates dominate biological assessment methods, because they are relatively easy to sample, and because of their widespread distribution and rapid response to pollution (ARMITAGE *et al.*, 1991). Also, because macroinvertebrates maintain their position in the river (by living on the stream bed or attached to aquatic vegetation), they best reflect the general quality of the water passing a particular stream reach (Anon., 1976; HAWKES, 1982; METCALF, 1989).

Lotic systems are characterised by highly variable numbers and species of invertebrates and usually quite discernible longitudinal patterns in the macro- and micro-distribution of this fauna (RABENI & GIBBS, 1980; MARCHANT *et al.*, 1985; BUNN *et al.*, 1986; WILLIAMS & MOORE, 1986).

Stream communities are adapted, in both their structure and function, to changing physical conditions of a river, from source to mouth and from season to season (VANNOTE *et al.*, 1980; MINSHALL *et al.*, 1983). The corollary of this is that changes in the physical, chemical or hydrological characteristics of a river are likely to induce corresponding changes in its fauna.

Substratum type is a major determinant of macroinvertebrate distribution and abundance (Cummins & LAUFF, 1969), because it is intimately associated with respiration, with the manner in which these organisms collect food (WILLIAMS & MOORE, 1986), with the quality of available food (GRAHAM, 1988), with the development of predator-prey relationships (PECKARSKY & DODSON, 1980; PECKARSKY, 1984), and with other behavioural adaptations such as net spinning (WILLIAMS & HYNES, 1973), attachment to river stones by threads, movement on the stream bottom and emergence of adult insects (WAGNER, 1984). Sediments may have particularly severe effects in biotypes such as riffles, where sediments may block up interstices between stones and reduce or eliminate the habitat of stream-dwelling macroinvertebrates, or where fine suspensoids may clog the gills of many of these animals.

The effects of sediments and suspended solids on benthic communities have been documented by many workers (*e.g.* CHUTTER, 1969; CHISHOLM & DOWNS, 1978; ROSENBERG & WIENS, 1978; PECKARSKY, 1984; MARCHANT *et al.* 1985; WILLIAMS & MOORE, 1986; GRAHAM, 1988). Although changes in their amounts are part of the natural seasonal variation in rivers (HARRISON & ELSWORTH, 1958; o'HOP & WALLACE, 1983), they are also a major form of pollution in lotic systems affected by urban and rural development such as road and bridge construction (BARTON, 1977; CHISHOLM & DOWNS, 1978; HORNER & WELCH, 1982; TAYLOR & ROFF, 1986; OGBEIBU & VICTOR, 1989), road use (SMITH & KASTER, 1983), dam construction (CHESSMAN *et al.*, 1987) and reservoir management (GRAY & WARD, 1982a, b). Experimental studies (Gammon, 1970; Rosenberg & Wiens, 1978; GRAY & WARD, 1982a) have also contributed to our knowledge of the effects of sediment and suspensoids on stream biota.

However, the literature sports widely diverging accounts of macroinvertebrate responses to suspended solids and sediment. CHESSMAN (1987) reported a noticeable increase in the amounts of suspended solids (from a range of 1 — 52, mean 7.9 mg/l, to a range of 5 — 520, mean 28.4 mg/l) during dam construction on the lower reaches of the Tanjil River in Australia. This was accompanied by a decline in density of some species, notably some Elmidae larvae, some Plecoptera, and Trichoptera, but no correlation between overall faunal density and proportion of silt in the substratum. Other measures of deterioration — a numerical similarity index and number of taxa — remained constant over the construction period. The study concluded that the low levels of sedimentation did not affect macroinvertebrate populations in a profound manner. ROSENBERG & SNOW (1975) and ROSENBERG & WIENS (1978) conducted experiments to quantify invertebrate drift in response to addition of suspended solids. Invertebrate drift occurred at concentrations of

30 mg/l suspended solids, but did not exceed 2% of the standing crop in 5 h, and was greater in autumn than summer. They concluded that sedimentation, not suspended solids was responsible for "most changes in a community". GAMMON (1970), on the other hand, reported a linear decline in invertebrate numbers (45—70% reduction in total numbers) with increasing suspended material (from 15 mg/l to 40 mg/l). GRAY & WARD (1982a) studied the effects of flushing of reservoir sediments on the North Platte River, Wyoming, to assess *inter alia* the relative contributions of suspended solids and settled sediments to macroinvertebrate abundance. They found that densities of five taxa chosen for study were highly correlated with increased suspended solids (a 20-fold increase with a peak of 442 mg/l), whilst there was no alteration of bottom substratum. SMITH & KASTER (1983), studying the effects of rural highway runoff on a third order hardwater stream, recorded suspended sediment concentrations of up to 567 mg/l (peak concentration during snowmelt), yet a similar number of taxa and average faunal numbers above and below the disturbance. They attributed the differences in density, biomass and richness between sites to differences in physical stream parameters *i.e.* current velocity and substratum.

BARTON (1977), working on the short-term effects of highway construction on Hanlon Creek, Ontario, found that suspended solids increased from a mean weekly level of 2.8 mg/l to 352 mg/l, with a rise in deposition of inorganic sediment from 0.06 $g/cm^2/d$ to 0.61 $g/cm^2/d$, with the highest levels occurring after final re-routing of the stream at the end of construction. He concluded that although there was a resultant decline in fish density during but not after construction, there was "no appreciable difference in benthic invertebrate numbers" or diversity. Notwithstanding some changes in community structure, population returned to normal in eight months (BARTON, 1977), once construction was complete. Barton's findings were contradicted by TAYLOR & ROFF (1986), examining the long term effects of construction on Hanlon Creek, the same subject of BARTON's (1977) study. They found continual changes in community structure for more than five years after completion of construction, these changes being attributable to ongoing high levels of suspended solids (25—95 mg/l). Suspended solids were flushed out of pools in which they had accumulated, despite dredging of these pools after construction. CHISHOLM & DOWNS (1978) examined recovery of communities after relocation of a stream channel and recorded population stabilisation two years after activities had ceased (determined by generic and diversity counts). However, community structure changed, reflecting a dominance of Trichoptera in the newly "recovered" community (CHISHOLM & DOWNS, 1978).

TAYLOR & ROFF (1986) found that the responses of macroinvertebrates to sediments were complex, and suggested that this was likely to be the case elsewhere, with local and regional variations. This may help to explain the lack of uniformity in results obtained by different workers (TAYLOR & ROFF, 1986). While this is instructive for those who would look for trends that may be used in applied conservation management, there are also a number of other more subjective, methodological constraints that impinge on the reliability or comparability of results.

One of these is the basis on which faunal community "health" is measured. Whilst many studies have favoured the use of community structure to quantify the effects of perturbation, these may be divided into two approaches (WASHINGTON, 1984). The first orders samples according to the overall similarities of taxa (ordination), for comparison with major environmental factors (including pollution). The second attempts to combine data on the abundance within species into a number, or index.

The majority of sediment studies have used a numerical index of diversity or an index of similarity, combined with taxon counts, total invertebrate numbers or density (BARTON, 1977; CHISHOLM & DOWNS, 1978; SMITH & KASTER, 1983; TAYLOR & ROFF, 1986; CHESSMAN, 1987; OGBEIBU & VICTOR, 1989). Diversity indices are based on the theory that a high diversity, with a fairly even distribution of individuals between species is indicative of undisturbed, stable environments (GHETTI & BONAZZI, 1977; HELLAWELL, 1977). Biological monitoring programmes based on measuring diversity changes assume that most forms of stress (pollution) reduce the complexity of the aquatic ecosystem (CAIRNS & DICKSON, 1971), and thus destroy the stability of the system. The notion that diversity begets stability has long been a matter of debate (WASHINGTON, 1984). These numerical indices have some inherent weaknesses, the most important being the loss of significant biological information because the identity of the taxa contributing to the community structure is not taken into account (LAWRENCE & HARRIS, 1979). Therefore, whilst they are usually able to indicate structural change, they cannot provide an indication of deterioration or rehabilitation of a system. Indeed, ARCHIBALD (1972) and HAWKES (1982) noted that low diversity may result from severe physical conditions in an *unperturbed* system. Thus although they provide information about change based on the relation between abundance (numbers of individuals) and species richness, their value cannot be interpreted as an indication of the direction in which change is occurring.

Seasonal variation in all these measures (diversity indices, numbers, generic count) may completely mask spatial patterns (MURPHY, 1978; WASHINGTON, 1984).

Considerable longitudinal change in rivers is also commonplace (HELLAWELL, 1977; HAWKES, 1982), because of changes in the suite of physical, chemical and physiographical features that determines the nature of the biota. One cannot assume that two sections of a river will necessarily display the same characteristics, even if they are situated close to each other. In order to study the effects of natural phenomena or anthropogenic activities on river systems, it is necessary to have a reference point, either spatially or temporally, to act as the basis on which to determine change (HELLAWELL, 1977). It is necessary to verify the similarity of faunal communities or assemblages at these "control" and affected sites, either before the perturbation occurs or, more usually, by establishing similarity between the physical features at both sites (*e.g.* altitude, gradient, flow and discharge minima and maxima). River sites grouped in this way generally correspond to faunal "zones" (HARRISON & ELSWORTH, 1958; HARRISON, 1965; KING, 1981; O'KEEFFE *et al.*, 1989).

However, the establishment of "control sites" for comparison of sediment effects has often been done without testing for temporal and longitudinal variation in community structure, as was the case in the studies of SMITH & KASTER (1983) and BARTON (1977).

Whilst similarity indices used as indicators of pollution have the above problems, their use in classification procedures such as community ordination techniques and cluster analysis appears to avoid some of these more serious drawbacks (HELLAWELL, 1977; GHETTI & BONAZZI, 1977; RABENI & GIBBS, 1980; HAWKES, 1982; SHEPARD, 1984; WASHINGTON, 1984; WRIGHT *et al.*, 1988). A community ordination technique compares the actual species composition of areas under study. This technique can be used to group samples along resource axes. It arranges benthic communities on a two axis system on the basis of differences in composition, as expressed by similarity coefficients (RABENI & GIBBS, 1980). Cluster dendrograms show the degree of similarity between sites. Such multivariate statistical techniques have proved to be a useful tool in the analysis of macroinvertebrate benthic data (KING, 1981; TOWNSEND *et al.*, 1983; COSSER, 1989) and in the development of programmes to assess and predict the biological quality of lotic systems (ARMITAGE *et al.*, 1990; ARMITAGE *et al.*, 1991). This technique can also be employed in pollution impact assessment if suitable control sites (spatial or temporal) are available to use as a baseline for comparison, and thus was used to study the effects of high concentrations of suspended solids on the macroinvertebrates of the Lourens River in the south western Cape.

The Lourens River is affected by farming activities for most of its middle and upper reaches. Over the past two years, intensified development of farming activities has required the draining of a large wetland area on the southern boundary of the Lourens River. Land drainage at Vergelegen Estate (A. M. Farms) has been achieved by a lattice of canals which were established in 1989 as part of a five-year development programme for A. M. Farms. The network of canals drains excess water into the Lourens River at four runoff points.

The canals are cut simply into the substratum, which comprises soil and small stones. Most of the banks are not yet covered by vegetation and thus are subject to severe erosion. In winter these canals drain large volumes of water. In 1990, for instance, the current speed along the length of a canal was high enough to wash away a farm road traversing such a canal (Mr Theron, Vergelegen Estates, pers. comm.). High runoff as a result of burst irrigation pipes and other technical problems also results in large volumes of water flowing through the canals and into the river, which has caused considerable clouding of the river water as far as the lower river (M. Peters, Chair, Lourens River Conservation Society, and C. J. Chorlton, riparian owner, pers. comm.), presumably reflecting a pulse of silt.

In late autumn 1991, two of these canals, which drain most of the land under agriculture, were modified to pass through "silt ponds" before emptying into the river (Figure 1). The objective in creating the ponds was to slow the water current, and thus to cause the settling out of much of the suspended load in the water before it emptied into the river.

Many riparian owners in Somerset West attribute perceived increases in silt concentrations to land drainage practises at Vergelegen Estates (see Appendix 1). Besides the aesthetic disturbance that this creates, river water is extracted below Vergelegen to supply drinking water for the town of Somerset West.

The aim of this study was to examine the impact of these canals and the silt ponds: to quantify pollution by suspensoids in the Lourens River, and to determine quantitatively and qualitatively some of its effects on the river ecosystem, through examination of the macroinvertebrate community inhabiting these reaches.

STUDY AREA

The Lourens River rises in the deeply incised Nuwejaarskloof of the Hottentots Holland Mountains, 60 km east of Cape Town, at an altitude of approximately

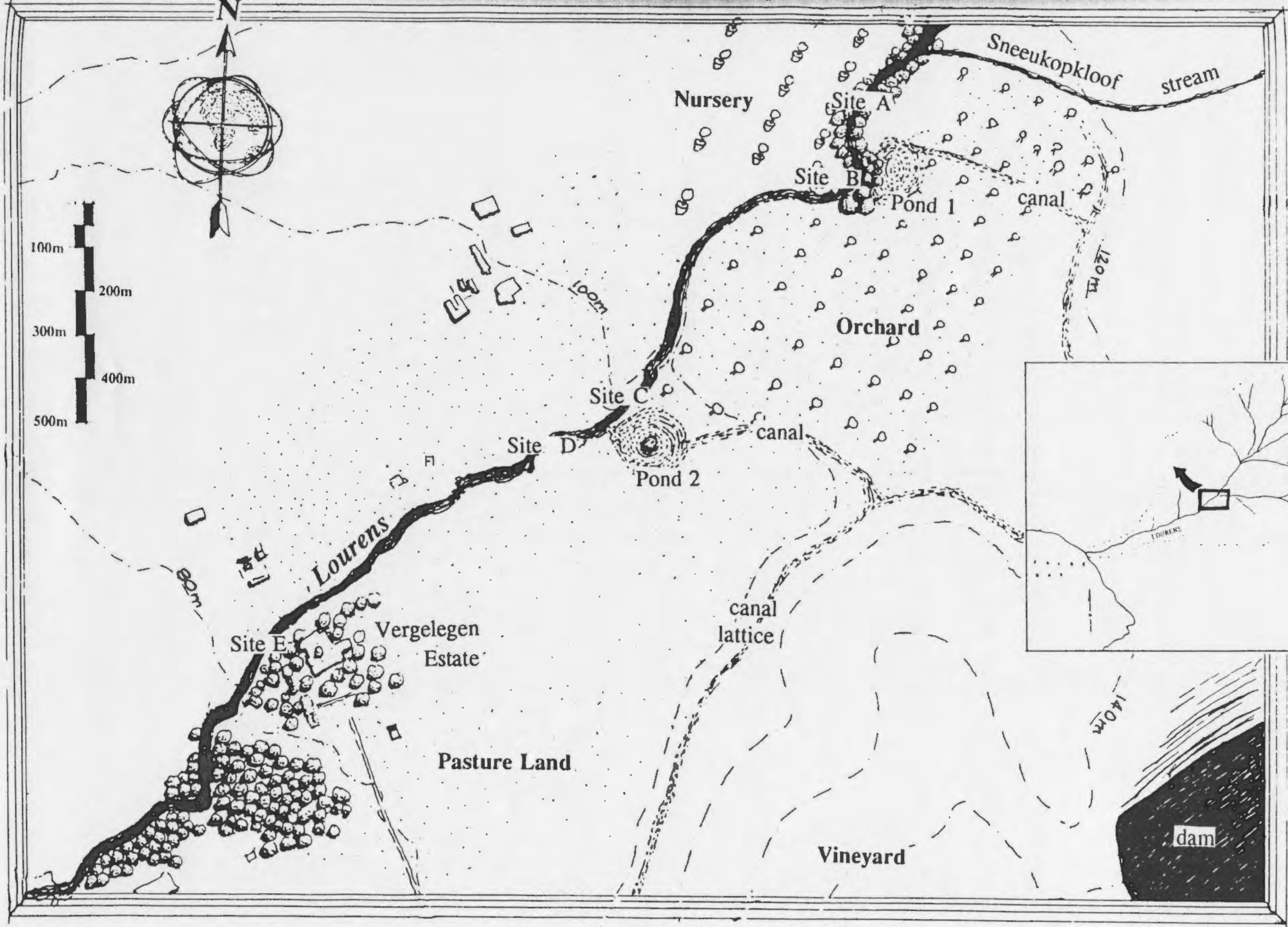


Figure 1. Map of the Lourens River, showing winter sampling sites and silt ponds.

1000 m, and is supplemented by streams arising in the Watervalkloof, Sneeukopkloof and Landdroskloof (HEYDORN & GRINDLEY, 1982; BEAUMONT *et al.*, 1986) (Figure 1). The river is one of a group of clear to dark, acid Cape rivers flowing over Table Mountain sandstone (O'KEEFFE *et al.*, 1989) in an area characterised by high winter rainfall and warm, dry summers (KING, 1981). Lourensford Estates recorded an average annual rainfall of 915 mm between January 1917 and December 1981, with a maximum in 1977 of 1470 mm and a minimum in 637 mm in 1934 (HEYDORN & GRINDLEY, 1982). A number of small dams of varying capacities have been constructed in the catchment area for the irrigation of agricultural lands adjacent to the river.

The study area was restricted to a 12 km stretch of river, from high in the mountains to the coastal plain. The river bed consists of sandstone boulders, and the river banks are formed from alluvial sand, silty clay and boulders (BEAUMONT *et al.*, 1986). In the middle and lower reaches, the banks are extremely vulnerable to erosion during winter flows, unless protected by vegetation or artificial constructions (BEAUMONT *et al.*, 1986; pers. obs.).

The most upstream section of the studied area courses through state and private land, part of which falls within the Hottentots Holland Reserve, and which supports pine forests interspersed with areas of fynbos. The latter forms a broad band alongside the river and extending down into the valley for some 4 km. The stream is steeply graded — the average gradient is 65 m/km (1:50 000 topographical map) and the mountain stream zone extends to an altitude of about 200 m (pers. obs). A landslide at Landdroskloof in 1990 altered the channel morphology markedly above the 280 m contour line, widening the stream and removing streamside vegetation. Summer stream flow was thus reduced to a trickle in this region, percolating through a rock and stone bed. Aside from this area, the stream bed of this zone consists of riffles of varying depths and including large boulders, alternating with pools, waterfalls and runs.

Downstream, the river and the Sneeukopkloof tributary form a natural boundary between the farms Lourensford Estates and Vergelegen (A. M. Farms). It is on the latter of these farms that an intensive development programme has required the draining of a large wetland area on the southern boundary of the Lourens River in order to establish conditions suitable for orchards and vineyards. Much of the orchard established on the drained wetland was in its first or second year of development in 1991 (Mr Theron, pers. comm.).

The river in this reach has a gradient of about 17 m/km (1:50 000 topographical map), and an average width of close to 6 m in summer and 10 m in winter. The substratum consists mainly of long stretches of shallow riffles, with small pools no deeper than 0.6 m and an average depth of 0.13 m in summer. In winter the average depth is 0.32 m, and the substratum consists mainly of deep and shallow riffles, and deep runs that carry fast-flowing water.

The lowest reach studied is an 8 km stretch of river coursing through the town of Somerset West and reaching the sea via the estuary at Strand. The substratum consists of small stones and pebbles on coarse sand, with patches of bedrock (HEYDORN & GRINDLEY, 1982). Marginal vegetation is dense (pers. obs.) although much of this has been deliberately cultivated as a bank erosion protection measure.

METHODS

Study sites

Most rivers exhibit longitudinal zonation. HARRISON (1965) has described these for South African rivers. Seven sites along the river were chosen for sampling in late February and early March 1991. These sites (labelled Sites 1 to 7) were chosen according to their physiographic characteristics, which are summarised in Table 1, in an attempt to establish the zonation of the Lourens River and to establish if, on the basis of physical, chemical and faunal characteristics, the river passing the silt pond outlets was encompassed within one zone.

Table 1. Description of summer sampling sites aong the Lourens River.

Sites	Distance from source (km)	Slope (m/km)	Altitude (m above sea level)	Description
1	2.6	80	340	Above all farming activity; forest plantation & fynbos
2	4.7	50	230	
3	6.1	22	175	Forest and orchard, also piggery and sawmill; land drainage practises; agricultural development
4	8.3	19	120	
5	11.0	15	80	
6	12.3	11	60	Residential areas
7	14.7	9	40	Centre of Somerset West

Extensive channel modification has been undertaken, either as part of flood prevention measures (M. Peters, pers. comm.) or in the building of farm roads across the river (pers. obs.). The river bed has been bulldozed or altered at Sites 3, 6 and 7, and is markedly wider and shallower at Site 3 than other neighbouring areas. A road crosses the river just below the sampling site. Sites 6 and 7 have been restructured considerably, in an attempt to confine flood discharges to the stream channel. The effects these measures can be seen by the erosion of the bank at Site 6, despite repeated rebuilding.

Silt loads from Vergelegen only affect the Lourens River when carried there by increased runoff. Whilst burst pipes and the like may carry a load of silt into the river, the effects of silt are likely to be most noticeable in winter, because consistently high runoff at this time of year can be expected. The main body of fieldwork in examining the effects of the canals and silt ponds was thus conducted in winter.

The winter sampling sites (Sites A—E) were chosen to allow comparison of conditions above and below the silt pond outlets. They are indicated in Figure 1, and described in Table 2. The control site (Site A) was in the same position as Site 4 in summer, and above Silt Pond 1. This pond has its deeper end at the water outlet, where the effluent flows overland for some 50m before entering the river. Site B, 500 m below the control and 100 m below the Silt Pond 1, was chosen to examine the immediate impact of suspended sediment from this pond on the fauna. Site C was situated 1 km downstream, and it was thus adjacent to Silt Pond 2, but upstream of its outlet. This site was established some distance from Site B, in order to monitor how far downstream the potential impact of suspensoids from Pond 1 extended. Pond 2 was larger and slightly shallower than Pond 1, and close to the edge of the river, although its banks were raised approximately 2 m above the river. At one end, the pond wall was lowered slightly, so that the effluent flowed from here over a bed of small stones, down into the river. Site D was established some 100m below this outlet, again to record the immediate impact, of Pond 2, in this case. It was also here that faunal samples were taken in winter, 1986 (BEAUMONT *et al.*, 1986), which could serve as a basis for comparison of changes in the fauna over time. Site E was established in the same position as Site 5 of the summer sampling sites, and was 600 m below Site D. The site was chosen to indicate if any recovery of the fauna was occurring, in the event that the silt ponds were having a deleterious effect on the river fauna upstream.

Table 2. Description of the winter sampling sites along the Lourens River, with reference to silt ponds at Vergelegen Estates.

Sites	Distance from source (km)	Slope (m/km)	Altitude (m above sea level)	Description
A	8.3	19	120	Control site, above silt ponds
B	8.8	17	120	Below outflow of Pond 1
C	9.8	15	95	Adjacent to Pond 2, above outlet
D	10.4	15	95	Below outflow of Pond 2
E	11.0	15	80	Possible recovery site

Sampling programme

Physical and chemical parameters were measured and the fauna sampled once at each of Sites 1—7 in summer, and Sites A—E on 3 July 1991 (representing winter). Physical and faunal samples were collected again on 14 August from Sites A—E because of the paucity of fauna in all the earlier winter samples. Both winter and summer samples were restricted to the riffle biotype.

Water samples from Sites A—E were taken weekly between 23 May and 14 August, and daily from 23 June to 29 June 1991, the latter period covering the largest spate during the study period. Samples were analysed for Total Suspended Solids (TSS), and the inorganic (ISS) components of TSS.

The efficiency of the silt ponds in reducing silt loads in the Lourens River was tested by measuring the TSS in the water flowing into the ponds from the canal system and water at the outlet.

Physical and chemical variables

At each site, on each visit, stream width was measured; depth and flow rate were measured at 0.3 m or 0.5 m intervals along a transect across the stream, and discharge calculated by summing the incremental discharge values along this transect. As the stream was not gauged, no continuous flow or discharge data were available.

Substratum type was estimated according to the percentage cover of sand (>1 mm diameter), gravel (>10 mm diameter) and stones (<100 mm diameter). A 0.25 m^{-2} metal grid, subdivided into 36 squares, was used for this process. The estimates of cover were made for each square, and these were summed to produce an estimate of percentage cover. Four replicates were taken within the riffle biotype at each site.

Measurements of conductivity (Crison CDTM-523 Conductivity Meter, standardised to 25°C), pH (Beckman Portable pH Meter), dissolved oxygen (Syland Portable Oxygen Meter, compensated for altitude) and temperature (mercury thermometer). were taken in the field. Probes were left in the stream for 30 minutes to "acclimatise" before measurements were made. A series of readings for each variable was taken across the width of the river. Dissolved oxygen was expressed as percentage saturation to facilitate comparison between sites.

Two one-litre water samples were taken at each site. One sample was processed at the Bellville Municipal Water Purification Laboratories and tested for nitrates (NO_3-N), ammonia (NH_4-N), orthophosphates (PO_4-P), chemical oxygen demand (COD) and total alkalinity, using the standard SABS method described by the American Public Health Association, the American Water Works Association and the Water Pollution control Federation (FRANSON, 1980). The second water sample was filtered immediately upon return to the Freshwater Research Unit laboratory, or frozen for later filtering and processing. Samples were filtered through pre-ashed, pre-weighed Watmann GF/F (0.45 μm pore-size) glass microfibre filters, dried at 60°C for 48 hours and weighed to determine TSS (total suspended solids per litre water). Further heating for four hours at 450°C in a muffle furnace was conducted to obtain the amount of Inorganic Suspended Solids (ISS).

Plate 2. The main drainage canal showing plant growth in summer. Periodic dredging in winter prevents overflow.

Plate 3. The lateral canal draining into Pond 1. Superficial plant growth in summer does not prevent erosion of the banks in winter

Plates 4. & 5.
Canal bank
erosion.

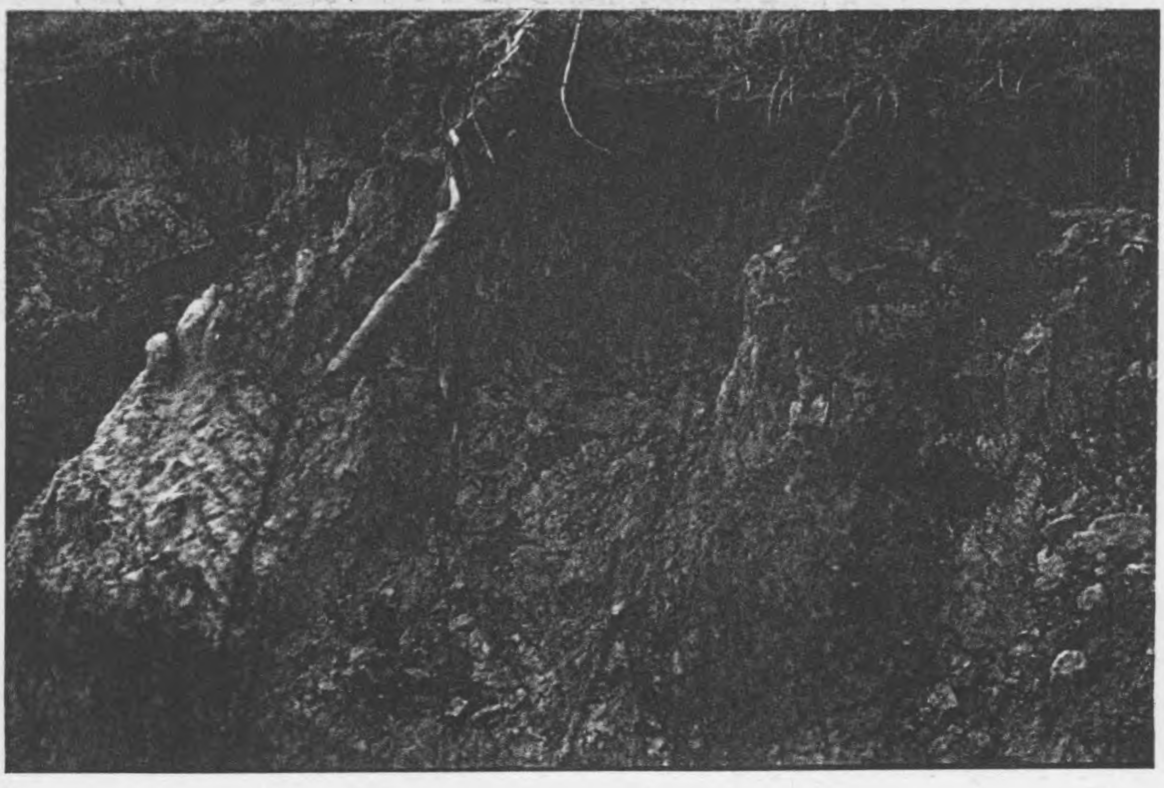


Plate 6. A canal with established bank vegetation is less susceptible to erosion.

Benthic fauna

Sampling of macroinvertebrates was restricted to stoney riffles. The collecting apparatus was a square-framed sampler designed by KING (1981), that sampled 0.1 m of stream bed. All moveable stones inside the frame were lifted and gently brushed to remove animals, visually checked, and placed outside the frame. Substratum to a depth of 10 cm was agitated to release buried animals. The downstream (collecting) side of the box was fitted with a funnel of 80 μm mesh netting with a detachable collecting jar at the end. Samples were immediately placed in 10% formalin, buffered with chalk, and transferred to 70% alcohol in the laboratory within seven days of collection. Six replicate samples were taken at each site, but because of time constraints on the scope of the project, only three were used for analytical purposes.

Summer samples were sorted for one hour with the naked eye, to remove the easily visible fauna, and then random subsamples were sorted under a 20X dissecting microscope for a period of two hours. Winter samples were fully sorted. Animals from the winter samples were identified to species level where possible, using the keys of PENNAK (1978), VAN EEDEN (1960) and a variety of unpublished new keys for South African invertebrates. Summer samples were identified as far as to family level in the case of Diptera, and to order for all other taxa, again, because of the constraints imposed by the scope of the project, and also because this level of resolution was considered sufficient for zone demarcation.

Analysis of data

Differences in the concentration of both total and inorganic suspended solids were investigated at the five winter sites, according to date of sampling, using the Friedman Two-way Analysis of Variance by Ranks. The Mann-Whitney-U Test was used to compare sites to determine differences between pairs of sites. The effectiveness of the two silt ponds in retaining suspended solids was measured by Wilcoxon's Paired Samples Test for matched pairs (ZAR, 1984).

The number of taxa and of individuals per square meter was determined for each sample from July and August. Both data sets were tested for downstream differences in these simple abundance measures, using the Kruskal-Wallis Analysis of Variance for non-parametric data (distribution of fauna was assumed to be contagious) and the Mann-Whitney-U Test for comparison of two samples (ZAR, 1984).

The relationship between sites, according to their faunal community structure, was investigated using the classification procedures compiled by M. R. Carr (Plymouth Marine Laboratory, United Kingdom), using the Bray Curtis (1957) index of similarity. Data from each of the winter sites were pooled and averaged, and all animal abundance counts were log-log transformed before analysis. Summer data were analysed on a presence/absence basis.

Results of the classification were summarised by group average sorting, and depicted in a dendrogram plot. Similarity between sites was reflected on a two-dimensional axis using the multidimensional scaling (MDS) ordination technique. Sites were then grouped according to percentage similarity, in order to reflect zonation (summer) or differences above and below the silt ponds (winter). An analysis of similarity (Simper) was employed to examine the result of ordination. It produced a numerical comparison of the similarity and the dissimilarity of these groups of sites (determined from MDS). Dissimilarity is expressed in terms of the differential contribution of various taxa toward percentage dissimilarity between groups of species (or sites in the case of this study).

RESULTS

Summer data

Physical and chemical characteristics

The physical and chemical measurements of the Lourens River in summer represent only a "snapshot" picture, which must be assumed to be a point within a range of values for these variables (Table 3).

The upper sites exhibited a greater variability in depth than those lower down in the river, although modification of the stream in the lower reaches (*e.g.* by bulldozing) made possible natural trends impossible to identify.

Consistent increases in conductivity were observed along the river, with values at Sites 6 and 7 being between double and eight times the values obtained at other sites. Both pH and alkalinity increased downstream, as did nutrient levels, with the highest orthophosphatevalue at Site 4. The sensitivity of the nutrient analyses performed by the Bellville Municipal Water Purification Works was not high enough to detect concentrations below 0.5 mg/l, which would mean that only high concentrations, relative to those naturally found in these reaches, would be detected.

Table 3. Physical and chemical characteristics at Sites 1 to 7 (summer).
s.d.: standard deviation n.d.: nutrient concentrations not detected by test.

	SITE.1	SITE.2	SITE. 3	SITE. 4	SITE. 5	SITE. 6	SITE.7
Depth (m)	0.25	0.28	0.07	0.17	0.21	0.23	0.13
[s.d.]	[0.13]	[0.18]	[0.04]	[0.06]	[0.11]	[0.12]	[0.06]
flow rate (m/s)	0.07	0.06	0.02	0.10	0.09	0.07	0.06
[s.d.]	[0.03]	[0.05]	[0.02]	[0.06]	[0.09]	[0.09]	[0.04]
discharge (m^3/s)	no value	0.12	0.02	0.13	0.16	0.10	0.04
width (m)	no value	5.40	8.00	6.35	5.20	3.70	3.23
substratum: sand	6.13	0.42	3.78	2.05	7.26	13.30	8.28
(% cover) [s.d.]	[1.91]	[0.83]	[1.93]	[2.42]	[7.37]	[8.95]	[9.08]
gravel	11.79	14.83	12.92	14.90	15.31	3.82	0.28
[s.d.]	[8.47]	[8.33]	[5.19]	[8.15]	[6.35]	[7.09]	[0.56]
stone	82.18	85.14	83.30	83.44	77.50	82.64	91.44
[s.d.]	[10.45]	[7.15]	[6.75]	[9.44]	[5.98]	[14.24]	[8.69]
TSS (mg/l)	3.00	0.00	1.00	0.00	6.00	0.00	0.00
conductivity (mS/m at 25°C)	3.03	4.05	4.35	7.00	10.55	19.03	24.50
alkalinity (meq/l)	0.50	0.50	0.50	0.50	1.00	1.20	1.70
pH	5.70	6.30	6.30	7.30	6.61	6.97	7.25
temperature (°C)	15.40	17.10	20.30	20.70	21.10	22.83	19.35
C O D (mg/l)	110.00	15.00	0.00	0.00	0.00	19.00	8.00
dissolved oxygen (% saturation	72.79	82.38 - 90.90	84.16 - 93.54	82.08 - 82.83	95.72	85.33	87.72
NH_4-N (meq/l)	n.d	n.d	n.d	n.d	n.d	n.d	n.d
NO_3-N (meq/l)	n.d	n.d	n.d	n.d	0.14	0.13	0.05
Orthophosphate -P (meq/l)	n.d	n.d	n.d	0.28	0.12	0.03	n.d

Sediments and suspended solids

There was no continuous, directional change in substratum composition with distance downstream, except that the highest percentages of sand in the substratum were recorded at the lower river sites (Sites 6 and 7). Concentrations of suspended solids remained low down the length of the river.

Site 4 corresponds to the first (Site A) and Site 5 to the last (Site E) of the winter sampling sites. The similarity between these sites in summer is evident from Table 3, except that, on the sampling day, the TSS and percentage of substratum covered by sand was greater at Site 5.

Invertebrates and river zonation

Table 4 indicates the distribution of different faunal groups along the river, from the high altitude stream (Sites 1 and 2) to the coastal plain (Site 7).

Cluster analysis revealed a distinctive grouping of the river sites into upper (Sites 1 and 2) and lower (Sites 3—7) zones (Figure 2). Two-dimensional scaling (Figure 3) seemed to separate sites along the X-axis, as a reflection of longitudinal change. The distribution of invertebrates along the Y-axis is less clear, and does not appear to match the physical and chemical data. The

Table 4. Faunal community structure based on presence/absence of major taxa at summer Sites 1 to 7. indicates presence of taxon.

PHYLUM	CLASS	ORDER	FAMILY	SITE 1	SITE 2	SITE 3	SITE 4	SITE 5	SITE 6	SITE 7
Arthropoda	Insecta	Diptera	Simuliidae	**	**	**	**	**	**	**
			Chironomidae	**	**	**	**	**	**	**
			Blephaceridae	**	**					
			Rhagionidae				**	**		
			Chaoboridae		**	**				
		Ephemeroptera		**	**	**	**	**	**	**
		Coleoptera		**	**	**	**	**	**	**
		Plecoptera		**	**	**				
		Trichoptera		**	**	**	**	**	**	
		Megaloptera			**	**		**	**	
		Odonata			**	**	**	**	**	**
Crustacea		Decapoda			**					**
Mollusca		Pulmonata				**	**	**	**	**
Annelida	Oligochaeta					**	**	**	**	**
Platyhelminthes					**	**	**	**	**	**

sampling schedule revealed that the time of day at which samples were collected seemed to present a similar distribution. This does not affect the zonation result, but is nevertheless a noteworthy phenomenon. Notably, both methods of determining the relationship between sites (Figures 2 and 3) showed that the greatest similarity in faunal community composition occurred between Sites 4 and 5. Sites 3 and 6 flanked the former pair of sites, and the group (Sites 3—6) had an average similarity of 90.03% (s.d. = 5.14). It is apparent from these results that Sites 4 and 5 belong to the same river zone. It is also these two

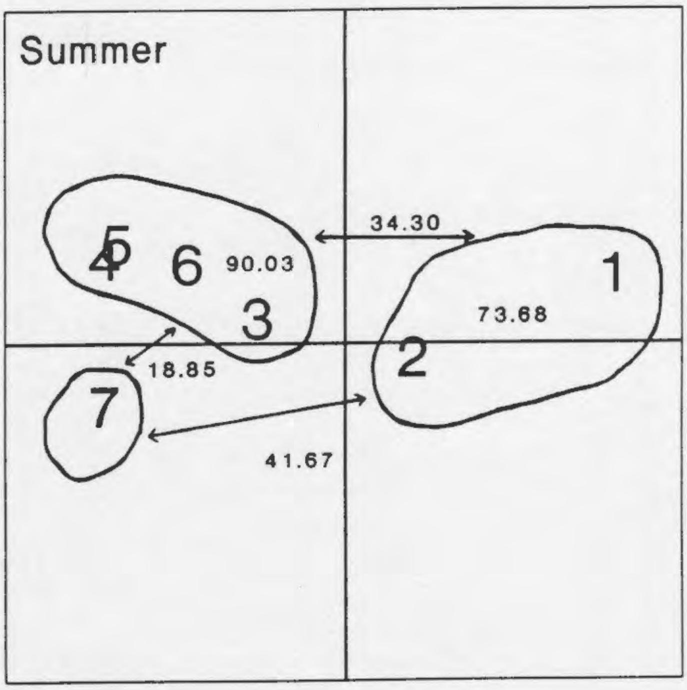


Figure 3. Two-dimensional ordination plot from MDS analysis of summer faunal data. The percentage similarity of groups is indicated inside cluster boundaries; dissimilarity between pairs of groups (ecological distance) is indicated by arrow-lines.

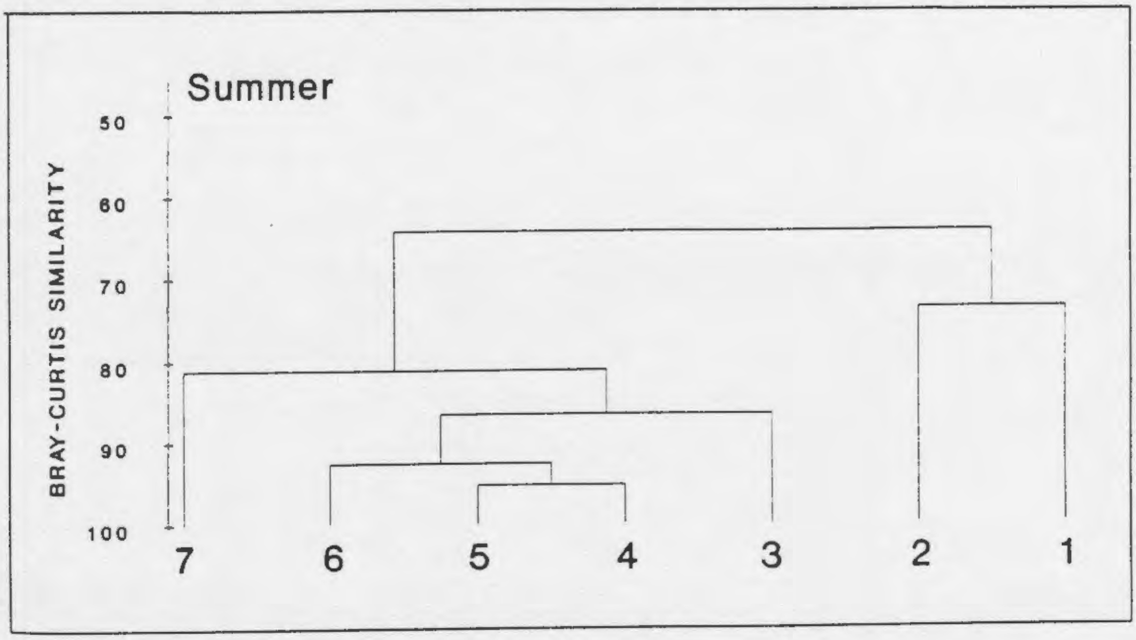


Figure 2. Dendrogram showing grouping of Sites 1—7 from cluster analysis of faunal samples, summer 1991.

Table 5. Physical characteristics of winter sampling sites A to E in June 1991.
s. d.: standard deviation.

14-08-91	SITE.A	SITE.B	SITE. C	SITE. D	SITE. E
Depth (m)	0.27	0.24	0.33	0.32	0.27
[s.d.]	(0.16)	(0.12)	(0.11)	(0.13)	(0.10)
flow rate (m/s)	0.35	0.39	0.44	0.35	0.39
[s.d.]	(0.17)	(0.15)	(0.21)	(0.18)	(0.11)
discharge (m^3/s)	1.48	1.51	1.80	2.03	2.39
width (m)	9.80	11.30	8.40	10.60	14.50
substratum: sand	5.60	3.51	5.94	2.36	2.81
[s.d.]	(4.31)	(3.66)	(5.05)	(2.39)	(3.83)
(% cover) gravel	7.53	4.51	0.83	6.84	4.56
[s.d.]	(3.84)	(2.8)	(0.71)	(2.94)	(0.47)
stone	86.88	91.98	93.23	90.80	92.64
[s.d.]	(5.33)	(5.98)	(5.58)	(3.74)	(3.36)
TSS range (mg/l) 8July - 14Aug	0-22	0-57	2-40	8-60	10-40

Table 6. Physical characteristics of winter sampling sites (A—E) in August 1991.
s.d.: standard deviation.

29-06-91	SITE.A	SITE.B	SITE. C	SITE. D	SITE. E
Depth (m)	0.45	0.27	0.38	0.34	0.30
[s.d.]	(0.20)	(0.10)	(0.15)	(0.20)	(0.13)
flow rate (m/s)	0.35	0.58	0.59	0.44	0.57
[s.d.]	(O.20)	(0.24)	(0.29)	(0.26)	(0.25)
discharge (m^3/s)	2.19	2.50	2.53	2.76	2.96
width (m)	8.80	11.25	7.20	10.15	11.75
substratum: sand	0.08	5.97	5.87	9.30	1.43
[s.d.]	(0.11)	(3.13)	(3.61)	(3.24)	(2.34)
(% cover) gravel	0.00	1.88	0.57	0.25	4.32
[s.d.]	(0)	(1.89)	(0.99)	(0.29)	(1.97)
stone	99.92	92.15	93.56	90.45	94.25
[s.d.]	(0.11)	(4.65)	(3.77)	(3.15)	(4.09)
TSS range (mg/l) 23May - 29Ju	6-216	8-253	9-214	12-233	11-200

sites that are situated above and below the silt ponds, and that contain all the winter sampling sites.

Winter data

Physical and chemical characteristics

Winter samples at Sites A to E (Tables 5, 6, 7) showed that besides variations in suspended solids (the chief variable under investigation in this study) and discharge, Sites A through E exhibited few differences. Sites A and C were somewhat deeper and narrower in mid winter than the average for the reach (Table 6) and Site B remained the shallowest (Tables 5 and 6). Chemical variables (Table 7) showed little fluctuation down the length of the river in winter. Discharge increased incrementally downstream on both sampling occasions, showing no sudden increase as a result of inflow of effluents from the silt ponds (Sites B and D).

Table 7. Chemical characteristics at winter sampling sites (A—E) recorded in June 1991.
s.d.: standard deviation n.d.: nutrient concentrations not detected by test.

	SITE. A	SITE.B	SITE.C	SITE.D	SITE.E
electrical conductivity (mS/m)	16.14	13.78	14.96	15.80	17.07
alkalinity (meq/l)	n.d	n.d	n.d	n.d	n.d
pH	6.30	6.30	6.50	6.50	6.50
temperature ($^\circ$C)	12.30	12.80	12.70	12.40	11.90
C O D (mg/l)	38.20	48.60	76.30	48.60	52.60
dissolved oxygen (% saturation)	95.37	89.64	92.70	88.35	94.80
NH_4-N (meq/l)	n.d	n.d	n.d	n.d	n.d
NO_3-N (meq/l)	n.d	n.d	n.d	n.d	n.d
orthophosphate -P (meq/l)	n.d	n.d	n.d	n.d	n.d

Sediments and suspended solids

Variables that may have been expected to show the effect of the silt ponds on the river, such as substratum type and conductivity, showed no discernible differences between Sites B or D (below the silt ponds) and the other sites.

The concentration of suspended solids (TSS), however, was the only variable whose values fluctuated apparently in response to the effects of silt loads from the ponds. Figure 4 shows weekly fluctuations in TSS over the winter study period, at each site, with the exception of week 3, when incorrectly processed samples were not used. Daily sediment concentrations during Week 5 are shown in Figure 5.

In general (Figure 4), all sites followed a similar pattern to that of Site A, which was not influenced by sediment run-off as a result of canalisation (although farming activities in general may increase substratum instability and erosion).

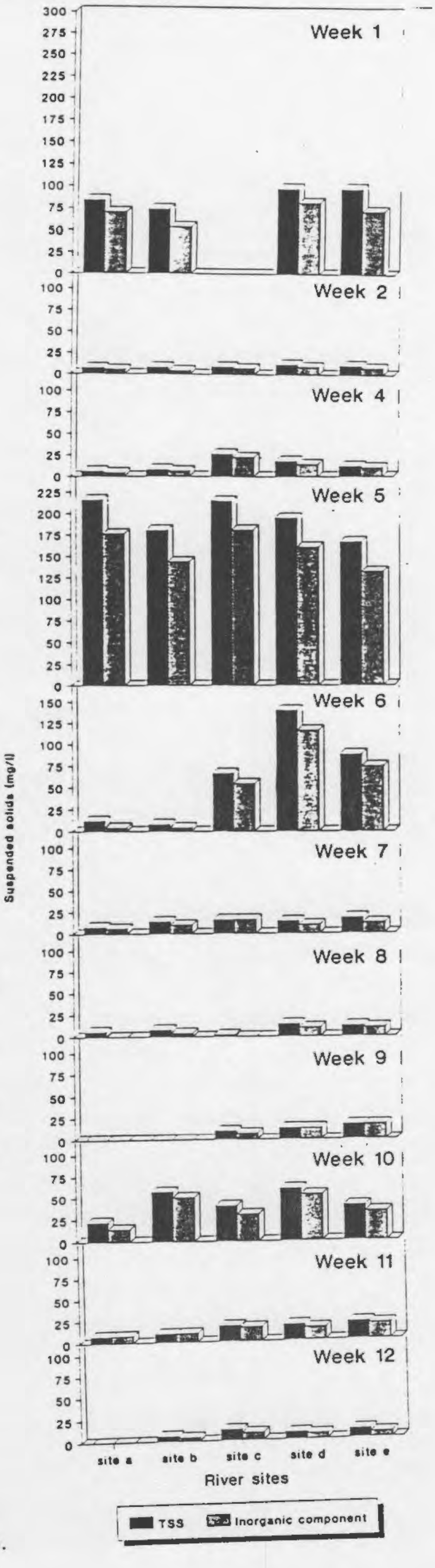


Figure 4. Weekly changes in TSS and organic suspended solids at each site (A—E) over Winter study period (week 3 values excluded).

At the end of spate activity, however, when the water levels were returning to base-flow levels and TSS concentrations falling (Site A, Figure 4), the pattern of TSS and Inorganic Suspended solids (ISS) at Sites B to D differed from that at Site A, Figure 4, Weeks 6 and 10; Figure 5). Sites B and D had fluctuating levels of suspended solids, which were for the most part mirrored by slightly lower values at Sites C and E respectively. During the period of the largest spate (Figure 5), Site B deviated from this pattern five days after peak flow (on 27 June, Figure 5), the TSS levels at Site B returning to concentrations close to that of Site A. In general, Site B appeared to have lower TSS concentrations than the sites below it (*e.g.* Weeks 6 and 11, Figure 4), and appeared to resemble Site A more frequently than did the other sites.

TSS concentrations did not follow a normal distribution, and hence non-parametric statistics were performed to investigate the apparent differences in TSS at the five sites over the study period.

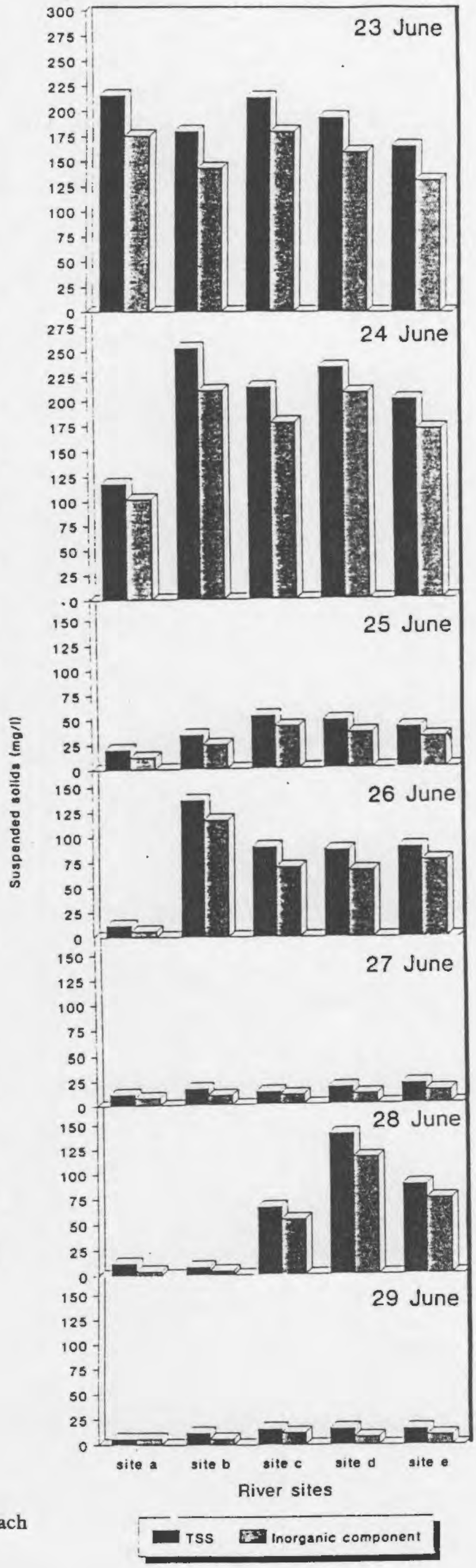


Figure 5. Daily changes in TSS and organic suspended solids at each site (A—E) over period of high discharge 23—29 June.

Two-way analysis of variance gave a highly significant difference (at 95% confidence levels) in both the total and inorganic component of suspended solids at the five sites (Table 8). Interestingly, the ranking obtained by the sites suggests a possible grouping of Sites A and B on one hand, and Sites C, D and E on the other (Table 8). In the case of ISS, the significance is greater and the "grouping" of sites on the basis of rank much clearer (Sites C, D and E share virtually the same rank) than in the case of the total sediment concentrations. Application of a Mann-Whitney-U Test shows Site A to be significantly different from Sites D and E, at the 95% confidence level, when comparing the TSS concentrations (Table 9). Examining differences on the basis of inorganic solids only isolates Sites C, D and E from Site A, and Sites D and E from Site B (Table 9).

Table 8. Ranks given to sites A to E on the basis of TSS and ISS concentrations. Test statistic and significance are shown for Friedman Test for differeneces in TSS, ISS over Sites A to E.

FRIEDMAN TWO-WAY ANALYSIS OF VARIATION BY RANKS		
Total Suspended Solids		
(n=16)	Site	Rank
	A	1.4688
	B	2.6563
	C	3.2813
	D	3.8750
	E	3.7188
Statistic:		25.3463
Probability:		P<<0.05
Inorganic Suspended Solids		
(n=15)	Site	Rank
	A	1.4000
	B	1.5333
	C	3.7000
	D	3.7000
	E	3.6667
Statistic:		26.1730
Probability:		P<<0.05

The effluent entering river from the Silt Pond 1, above Site B, had a significantly lower concentration of suspended material in (Table 10) than that of the drainage water entering the pond from the canal system. The same was not the case for the lower pond (Pond 2). Figure 6 appears on first

Table 9. Results of Mann-Whitney U Test to locate differences in TSS (left of table) and ISS (right of table) concentrations between Sites A—E. Significant differences between pairs of sites is indicated by silhouetted numbers.

MANN-WHITNEY-U TEST FOR DIFFERENCES BETWEEN SITES

(TSS)	SITE E	SITE D	SITE C	SITE B	SITE A	(ISS)
A	0.0147	0.0103	0.1006	0.2207		A
B	0.1281	0.1239	0.4786		0.1276	B
C	0.5686	0.4897		0.1162	0.0112	C
D	0.9243		0.7623	0.0335	0.0015	D
E		0.6784	0.91	0.0307	0.0015	E
	SITE E	SITE D	SITE C	SITE B	SITE A	

reading to show a large difference between inflowing and outflowing suspended solids from Pond 2, with an increase in the outflowing solids (*e.g.* week 10 in Figure 6). However, close comparison of the different samples in the graph reveals that the sediment concentration of the effluent water from Pond 2 fluctuated widely, bearing little direct relation to the amount entering the pond. Comparisons are only possible from week 6 onward, and perhaps trends would be more explicit with increased replication of samples (eight replicates were taken for statistical analysis). In contrast to this result, however, Pond 1 (the upstream pond) showed a clear pattern in the relationship between incoming silt concentrations and those of the effluent (Figure 6). There was a small, but noticeable and consistent decrease in total and inorganic suspended solids.

Table 10. Results of Wilcoxon Matched Pairs Test for differences in silt content of inflowing drainage water and effluent entering water at Pond 1 and Pond 2.
△ : significant difference at 95% confidence level.

EFFECTIVENESS OF SILT PONDS WILCOXON PAIRED TEST		Probability (P) by: signs	ranks
TSS (n=8)	pond 1	▽ 0.0771	▽ 0.0209
	pond 2	0.2888	0.2076
Inorganics (n=7)	pond 1	0.1306	▽ 0.0346
	pond 2	0.4497	0.4469

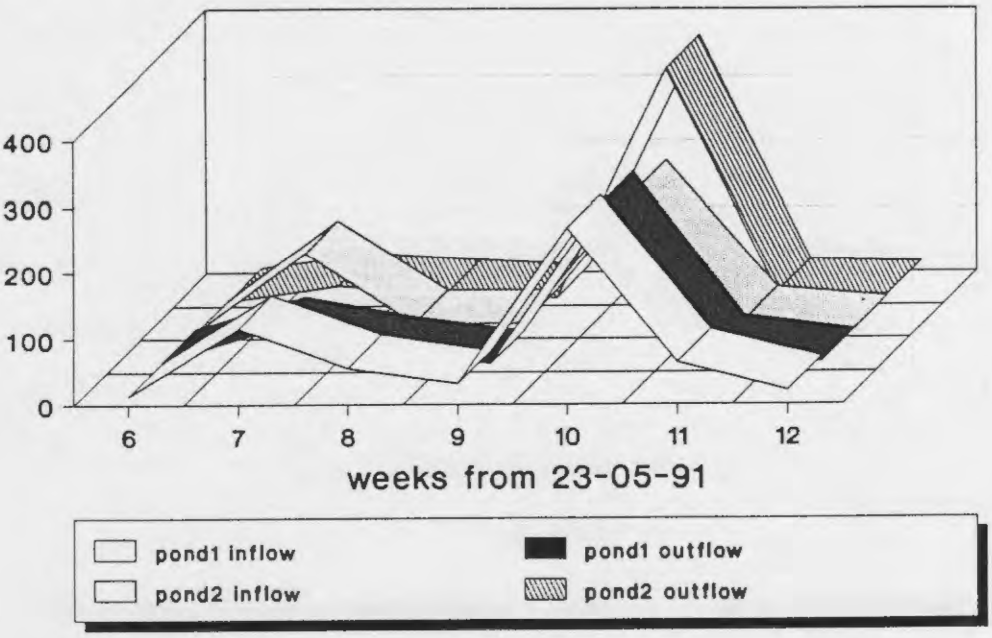


Figure 6. Fluctuations in TSS over the study period (27 June to 14 August) at inlet and outlet of ponds.

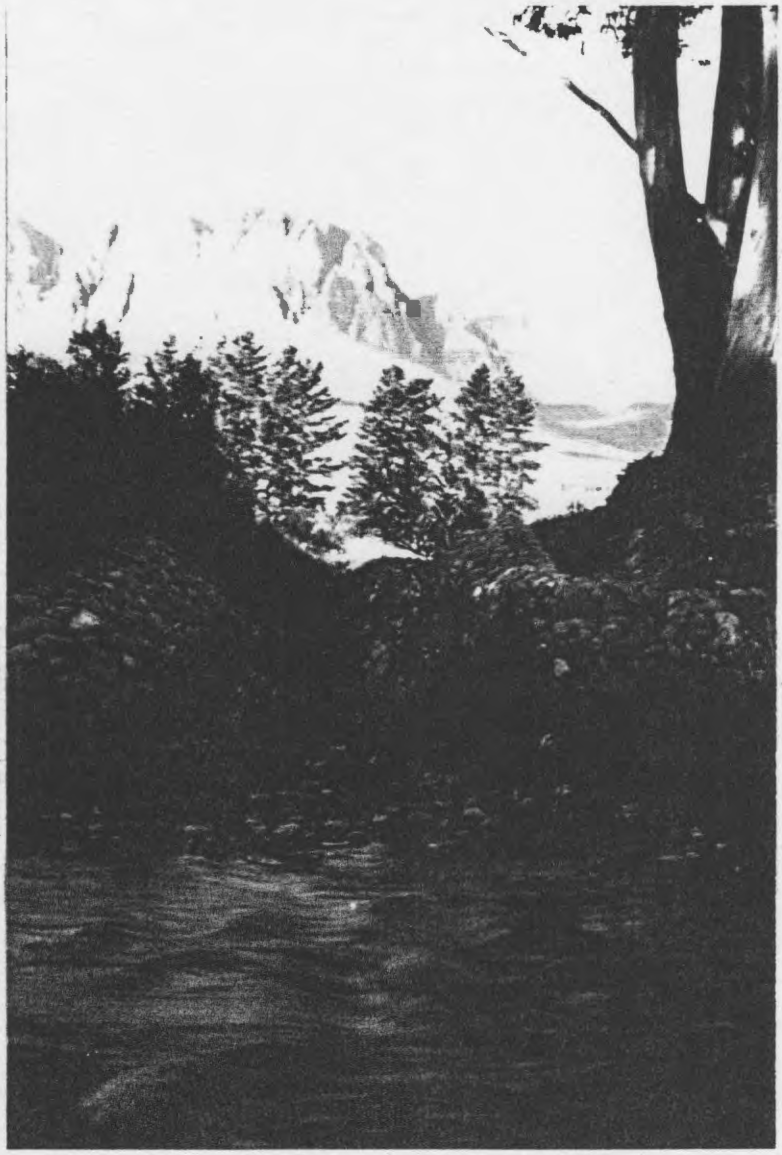


Plate 7 Initially canals emptied directly into the Lourens River. This canal was redirected in 1991 to Silt Pond 1.

Plate 8. Effluent from Silt Pond 1 flows overland for some 50 m before entering the Lourens River.

Plate 9. Silt Pond 2

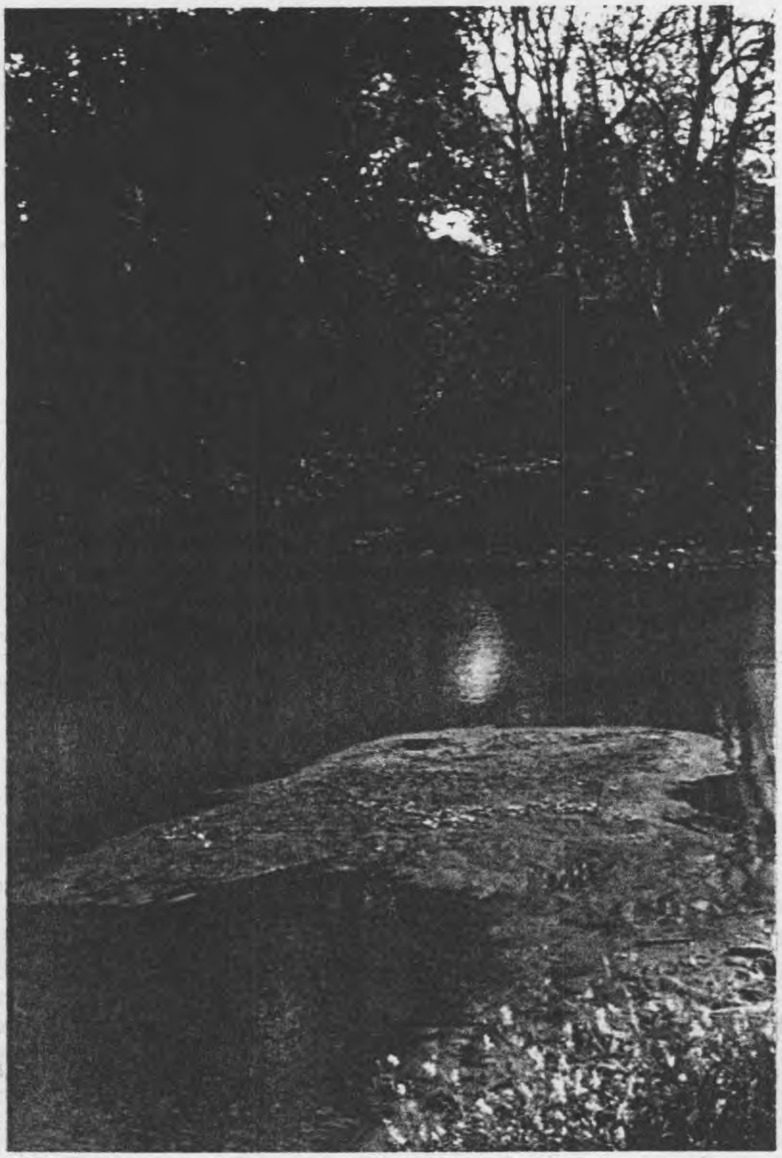
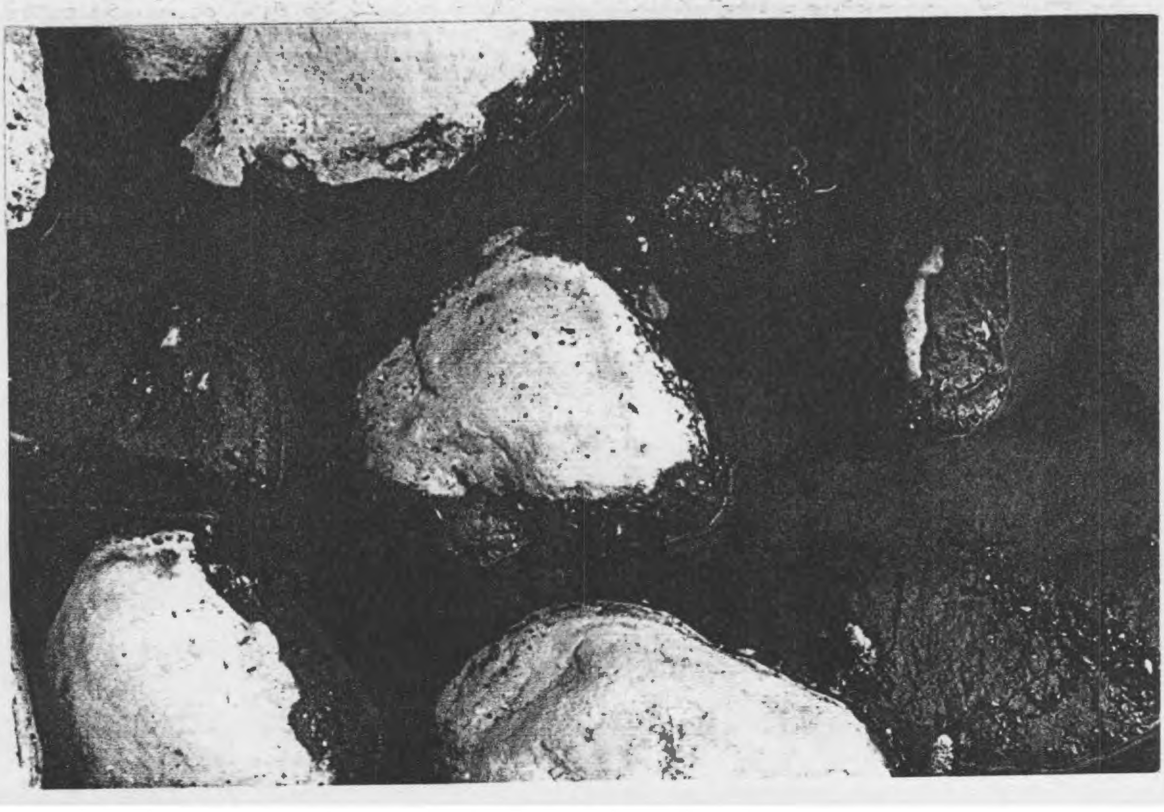


Plate 10. The settling out of suspensoids in drainage water as it enters Silt Pond 2.

Plate 11. Heavily silted stones where drainage water from Silt Pond 2 enters the Lourens River indicates a high silt load.



Effects of sediment on invertebrates

Simple abundance measures (Figures 7 and 8) did not establish any statistically significant changes in community structure below the silt ponds. July values for

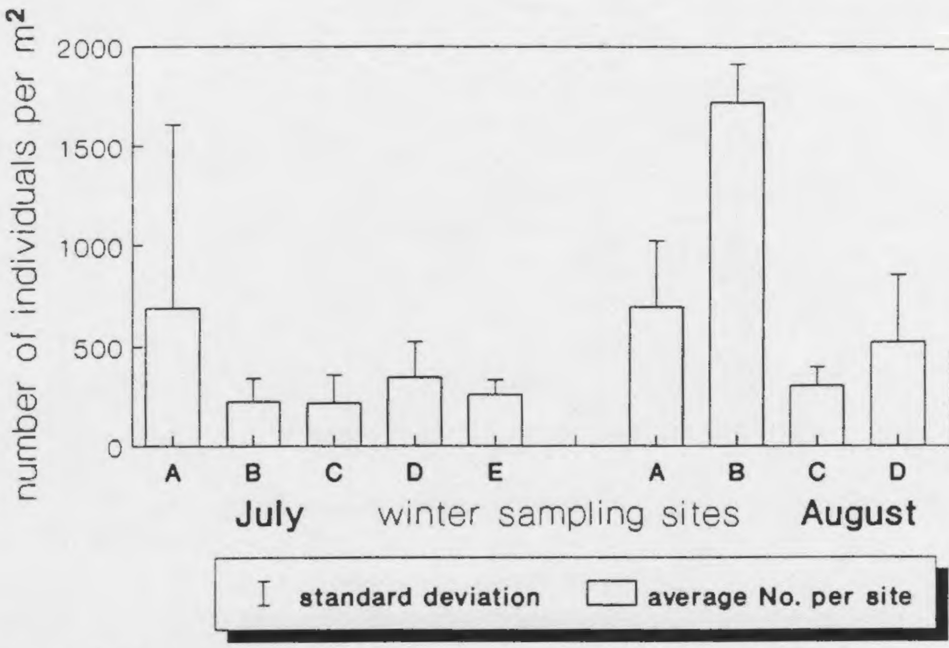


Figure 7. Number of individuals at Sites A—E, in July and August 1991. Numbers at each site represent an average of three replicates.

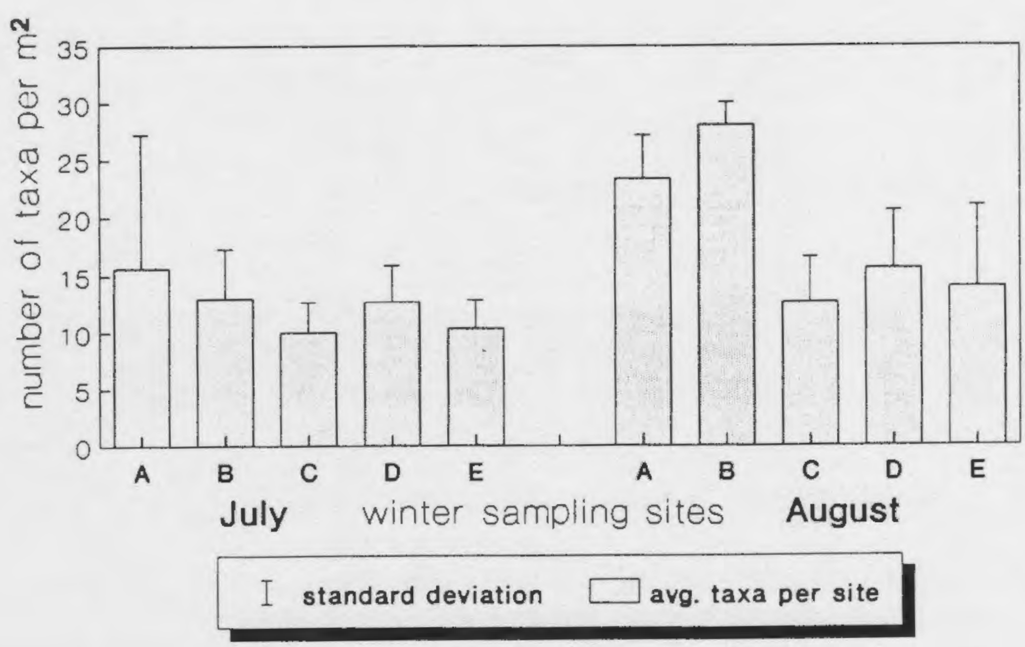


Figure 8. Average number of taxa at Sites A—E in July and August 1991. Numbers of taxa represent an average of three replicates.

faunal and taxon counts show no significant differences downstream, when all sites are compared (Kruskal-Wallis, $P >> 0.05$). In August there was a difference in the number of taxa between sites (Kruskal-Wallis, $P < 0.05$). However, The Mann-Whitney-U test failed to locate any differences between pairs of sites ($P > 0.05$).

Testing for differences in numbers of individuals or number of taxa at any one site from July to August gave no significant differences (Mann-Whitney-U Test, $P > 0.05$). This is a somewhat unexpected result, given, for example, that the number of individuals at Site B appears to increase dramatically over this period (Figure 7).

July data was not analysed using similarity measures, as it was felt that the samples did not adequately reflect community structure (see section on methods).

The results of cluster analysis of August data show the clear grouping of Sites A and B (Figure 9) which have

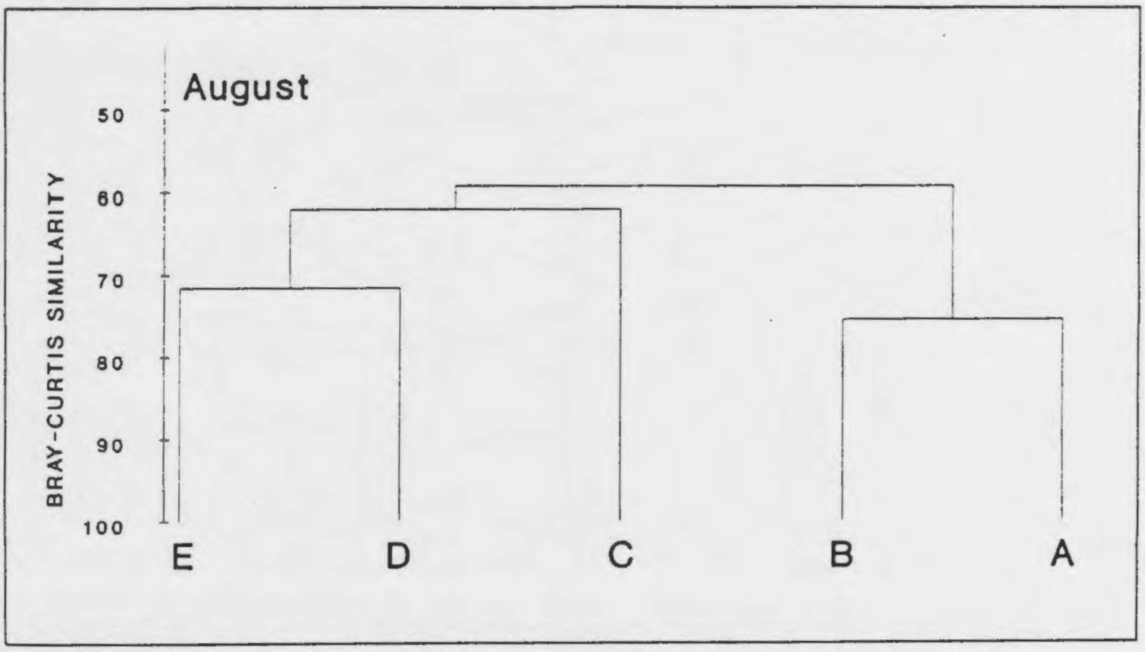


Figure 9. Dendrogram showing grouping of Sites A—E. From cluster analysis of faunal samples, August 1991.

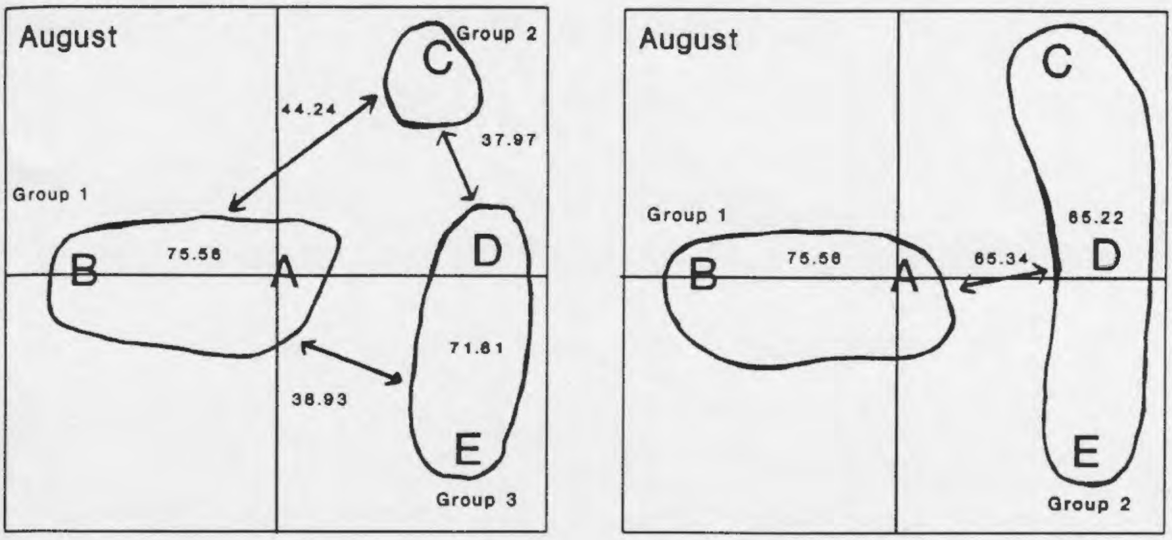


Figure 10. Two-dimensional ordination plot from MDS analysis of August faunal data. The percentage similarity of groups is indicated in side cluster boundaries; dissimilarity between pairs of groups (ecological distance) is inducated by arrow lines.
Figure 10a reflects three groupings of sites, chosen at a 70% similarity cut-off (Figure 9); Figure 10b reflects two groups of sites chosen to investigate the relationship of Site C to group of Sites A, B and D, E.

an average similarity of 75.56% (Figure 10a, b), calculated from the Percentage Similarity (Simper) Analysis. The ordination plot from MDS (Figure 10a, b) indicates a strong disturbance gradient along the x-axis, with Sites D and E being furthest removed from Sites A and B along this axis. Sites D and E together form a group with 71.61% similarity (Figure 10a), whilst Sites C, D and E are only 65.22% similar on average, with a standard deviation of 5.75% (Figure 10b). Figures 10a and 10b show the relationship between groups of sites where Sites C, D and E form a single agglomeration (Figure 10a) or where they form two groups (Figure 10b). In the case of the latter, *i.e.* where Sites D and E form one group and Site C forms another, the percentage dissimilarity between the three resultant groups (including the A/B group) is comparable, although Site C (in a group of its own), and the D/E group show the greatest affinity of all pairs of groups (dissimilarity = 38.93%, Figure 10a). It is clear from these figures that, whilst Site

C is closer in terms of overall community structure to the D/E group, at a 70% similarity cut-off point (used for summer data analysis) (Figure 9) this site forms a group of its own. Three groups were thus determined (Figure 10a) for examination of the taxa contributing most to dissimilarity in community structure.

The five taxa contributing most to dissimilarity between Group 1 (Sites A and B), Group 2 (Site C) and Group 3 (Sites D and E) are shown in Table 11, the result of the Simper analysis. It is worth noting here that the key taxa that distinguish Group 2 (Site C) from both remaining groups are Chironomidae, Coleoptera and Plecoptera. In contrast, dissimilarity between Groups 1 (Sites A and B) and 3 (Sites D and E) arises from major differences in Leptophlebiidae, Chironomidae, Trichoptera, Baetidae and Coleoptera. Table 12 reflects the complete faunal composition of the three groups identified by classification procedures. Taxa unique to one group are indicated, as well as those shared by different combinations of groups. Interestingly, Group 2 (Site C) and Group 3 (Sites D and E) do not possess any taxa unique to their union, whilst three taxa are shared by Group 1 and Group 2, with similar abundances (numbers of individuals in each taxon). The asterisks indicate

Table 11. Results of Simper analysis, showing the five most important taxa, in relation to their contribution to dissimilarity between groups of sites.
Group 1: Sites A and B
Group 2: Site C
Groups 3: Sites D and E

Groups	Taxa	% Dissimilarity contributed
1 vs 2	Chironomidae sp. E	5.44
	Chironomidae sp. CC	5.37
	Helodidae sp.A	4.82
	Elmidae sp. A	4.32
	Chironomidae sp. J	4.07
1 vs 3	*Lithogloea harrisoni*	4.01
	Chironomidae sp. G	3.69
	Athripsodes spp.	3.66
	Baetidae juveniles	3.6
	Elmidae sp. C	3.49
2 vs 3	Chironomidae sp. CC	7.03
	Plecoptera sp. A	6.16
	Chironomidae sp. G	5.06
	Chironomidae sp. E	4.89
	Chironomidae sp. J	4.64

the relative abundances (number of individuals) in taxa, in a coarse manner only, so as to facilitate comparison. Full lists of taxa and numbers

TAXON	GROUP 1 Sites A&B	GROUP 2 Site C	GROUP 3 Sites D&E
Athripsodes spp.△	**		
Leptoceridae sp.A△	*		
Orthotrichia spp.△	*		
Adenophlebioides spp.▲	*		
Aprionyx spp.▲	*		
Lestagella spp.▲	**		
Rhithrocloeon spp.	**		
Elmidae sp.C▲	**		
Noteridae adults	**		
Chironomidae sp.D	*		
Chironomidae sp.K	*		
Rhagionidae ▲△	*		
Dugesia spp.	**		
Chaoboridae spp.	*		
Hydracarina sp.D	*		
Acentrella spp.▲△	*	*	
Cheumatopsyche spp.▲△	*	*	
Chironomidae sp.G▲△	**	**	
Lithogloea harrisoni △	***	**	*
Baetis spp.	***	**	**
Baetidae juveniles	***	*	*
Plecoptera sp.B▲△	**	*	*
Chironomidae spp.	***	*	**
Hemiptera ○	*	*	**
Collembola	**	**	**
Hydracarina sp.A	**	*	**
Lumbriculidae	**	**	**
Naididae ○	**	**	***
Coryaldus sp.		*	
Castanophlebia calida	***		*
Elmidae sp.A	**		*
Helodidae sp.A	***		*
Plecoptera sp.A▲△	**		***
Blephariceridae sp.B▲△			*
Centroptiloides sp.			*
Elmidae adult			*
Helodidae adult			*
Hydraenidae adult			*
Dryopidae sp.A			*
Ferrissia spp.○			*

Table 12. Faunal composition of Groups 1 to 3, from cluster analysis

* rare
** present
*** abundant

▲ mountain stream fauna
△ upper river fauna
○ lower river fauna

for both July and August are shown in Appendices 2 and 3. In Table 12 taxa that tend to be characteristic of "mountain stream", "upper river" or "lower river" types, on the information of KING (1981) and HARRISON & ELSWORTH (1958), are shown.

DISCUSSION

Summer study reach

Physical and chemical characteristics

The Lourens River is highly modified in its upper, foothill and lower river zones by the management practises of farming, but particularly flood prevention.

Variability in depth in the Lourens River was the probable result of channel modification, which has been extensive as part of flood prevention measures (M. Peters, pers. comm.). The severe bank erosion upstream of Site 6 was implicated in the "low diversity" of the invertebrate fauna found by BEAUMONT *et al.* (1986) in a brief examination of the benthic fauna.

Flow rates and discharge values obtained in the study lose comparative worth, because they are less a function of natural physical features than of rates and volumes of water extraction down the length of the river. Water is extracted directly from the river and its tributaries for farming activities (HEYDORN &

GRINDLEY, 1982), as well as for the municipalities of the Strand and Somerset West at points above Sites 4 and 5, and at Site 6. Thus whilst catchment drainage would be expected to be reflected in increased discharge with distance from source, the lower sites indicated considerable attenuation of water supply to the river environment.

Nutrient analyses used could only detect changes on a scale far greater than the order of change usually found in the pure foothill zone waters. Any nutrients detected, therefore, must certainly have reflected organic effluents entering the river.

Low discharge in summer, coupled with periodic domestic activity such as clothes washing (pers. obs.) would cause fluctuations in water chemistry. The continuous farm run-off, carrying with it high concentrations of nutrients, accounted for the presence of greater quantities of dissolved ions and higher pH and alkalinity of the water as it flowed through the foothill zone, and hence the presence of shelled animals such as molluscs (Table 4) in this zone.

Fluctuations in the chemical characteristics and extensive changes to stream and bank morphology, and thus most physical features, in the foothill zone and the lower river were clearly indicated by the summer data. These are likely to destroy some of the distinguishing features of some, or widen the girth of other microhabitats that make up the different faunal zones.

Invertebrates and river zonation

Although the application of classification procedures (cluster analysis and multi-dimensional scaling) to summer faunal data would show better resolution had all the material been identified, some interesting patterns emerged that could be compared with abiotic characteristics of the river. The classification using group average sorting showed a distinct mountain-stream group, and then bracketed the remaining sites. These may be subdivided in a number of ways. The two-dimensional ordination plot facilitates decision-making in this regard, especially if used in conjunction with the Simper Analysis. In this study, they presented a visual picture of "ecological distance" (*i.e.* differences in distribution determined by the relationships between the different fauna and their environment) between sites, and a quantitative measure of the similarity or dissimilarity of clusters of sites, as shown in Figure 3. The ordering of sites along the x-axis to reflect longitudinal change showed that toward the lower reaches of the foothill zone, the sites became very similar. The close proximity of two of the groups, excluding Site 7, may be interpreted as suggesting that the foothill zone and lower river (where it is confined to rocky bed) were not well distinguished from each other. Zonation patterns are expected to be more obvious in summer (J. King, pers. comm.) because they are not subjected to the "scrambling" effect of high discharge and strong current speeds of winter conditions. However, water extraction, nutrient loading and alteration of stream configuration could widen the complement of species common to both zones, and restrict the presence of those more narrowly confined to a particular set of conditions. For example, molluscs occurred all the way upstream as far as Site 3, the top of the foothill zone. Interestingly, Plecoptera were present as far downstream as Site 3, despite the shift in some important characteristics — flow, discharge and temperature. Plecopterans are sensitive to organic pollution, temperature and flow regime (PENNAK, 1978), which explains their restriction to extend downstream only to the upper limits of the foothill zone. Thus although there were distinct zonation patterns with regard to the mountain-stream, increased sampling is necessary to define the boundaries between the foothill zone and lower river, because of the effects of human intervention.

Notwithstanding the above constraints, both the cluster dendrogram and the ordination plot (Figures 2 and 3) grouped Sites 3 to 6, with over 85% similarity. They also showed that the greatest similarity in faunal community composition occurred between Sites 4 and 5. It is therefore clear that these two sites were situated within the same zone. The grouping of Sites 3 and 6 with Sites 4 and 5 suggests that they formed part of the same zone.

Winter study reach

Physical and chemical characteristics

The uniformity of the stream reach between Sites 4 and 5 in summer is further emphasized by the results of winter sampling of the physical and chemical parameters governing Sites A—E (Tables 5, 6, 7). The importance of this observation for the study is that, notwithstanding the fact that faunal communities change considerably from summer to winter (KING, 1981; HARRISON & ELSWORTH, 1958), the high similarity between the winter sites (summer Sites 4 and 5) provides a basis upon which one can confidently compare the results obtained from examination of sediment impact on the fauna. Also, the zones identified in summer will remain the same in winter, even though the faunal communities will change (J. King, pers. comm.).

As has been shown from the results, the amount of sediments (sand) in the substratum in fact changed very little longitudinally or throughout the year, despite the increased TSS loads in winter (Tables 3, 5, and 6). This corresponds with the findings of GAMMON (1970) and GRAY & WARD (1982) that there is no direct relationship between the quantity of solids in suspension in a river and that which settles out on the bed.

Whilst some authors (*e.g.* CHESSMAN, 1987) suggest that depletion of river fauna is not correlated with the amount of sediment on the substratum, others (*e.g.* ROSENBERG & WIENS, 1978) hold it to be more important than suspended solids in relation to the effects on fauna.

Although there were no significant differences in sand cover from site to site, this variable is difficult to measure accurately, and the dynamics of silt movement in stone interstices is poorly understood. This study was not able to detect the possible presence of this fine settled sediments, which may nevertheless have had a significant impact on the riffle fauna.

In the context of the above, even small changes in depth may have considerable consequences for the micro-environment of the fauna, in the event of high TSS concentrations. Shallow water is likely to increase the effects of turbulence around each river stone, thus dispersing sediment particles from stone or gravel interstices. REYNOLDS *et al.* (1990) argue that depth, not current velocity is the controlling factor in determining the settling-out rate of suspensoids. They suggest that particles will become entrained in the overall motion, given a high enough turbulence, except in the boundary layer. Deeper water should have the effect of increasing the boundary layer as the rate of decrease of horizontal velocities increases towards stream bed. The result of this will be greater laminar flow and less turbulence.

In this respect Site B, with an average depth of one third to a half less than the other sites, may represent a more sediment-free micro-environment for benthic macroinvertebrates than for example Site A or Site C.

Changes in depth thus *may* reinforce the effects of high silt loads. More importantly, though, the uniform physical and chemical characteristics confirm earlier evidence of the zonal identity of Sites A and E. The only feature that shows differences between sites is that of TSS. It is reasonable, therefore, to conclude that other differences that correlate with TSS changes can be attributed to the differences in silt loads in this reach.

Suspended solids

High TSS concentrations at Site B and D occurred in pulses for a week after the storm on 23/24 June, and appeared to induce corresponding increases in TSS levels down the length of the river (Figure 5). Concentrations of silt and other suspended solids can be expected to vary quite considerably over time, as they are taken into suspension as a result of stochastic events such as storms, which in natural systems carry large volumes of silt downstream (o'HOP & WALLACE, 1983). The benthic fauna is adapted to survive these events by avoidance, however, seeking shelter for a limited period in the hyporheic zone (HYNES *et al.*, 1976). Continuous quantities of suspended solids in a stream are potentially more harmful, therefore, than a pulse during a spate, because the fauna is not able to survive in the hyporheos, which is outside of its natural feeding habitat, for long periods. Thus whilst the post-spate concentrations (Figure 4; Figure 5: 25—29 June) measured at Sites B, C, D and E do not match the "natural" load in a storm, their effect on the biota must be interpreted with reference to the timing of sediment pollution.

It is conceivable that the most critical period during which the fauna would require silt-free conditions is around the return to base-flow after a spate, for the reasons discussed above. On 24 June (Figure 5), it can clearly be seen that, although the TSS levels in the natural river are decreasing, those measured at sites affected by the silt ponds show continually greater TSS concentrations.

There is little information as to the exact time spent in the hyporheos by invertebrates before re-emergence onto the stream bed. However, at the time when some taxa were reappearing (28 and 29 June, pers. obs.), TSS concentrations were higher than 60 mg/l (Figure 5). Many authors have noted drift by benthic fauna upon exposure to suspended solids (*e.g.* KING, 1981; O'HOP & WALLACE, 1983). ROSENBERG & WIENS (1978) have hypothesized that pulses of invertebrate drift may occur as a result of the sequential repopulation of surface substrata by macrobenthos, from the hyporheos, and their subsequent drift to avoid continuing silt loads.

Clearly, then, if silt loads remain above the tolerance limits for the macrobenthos, recolonisation of stream reaches denuded by induced drift will not take place.

The similarity in the concentrations of suspended, particularly inorganic, sediments at Sites C, D and E, deserves some consideration. It is apparent from Figures 4 to 6 that Pond 1 does not have the same impact on suspended sediment levels at Site B as Pond 2 does on Site D. Not only is there a noticeable decrease in the TSS loads at Site B, relative to downstream sites, but statistical tests also indicate that Sites A and B have lower suspended solids concentrations than any of the other sites.

The efficiency of Pond 1 in reducing the sediment content of water at its outlet can partly explain this pattern. The pattern of sediment release from Pond 2 indicates that although sediment may settle out as it enters the pond from the drainage canals, this fine silt is resuspended easily with the onset of new rains. This process is shown in Figures 4, 5, and 6 with reference to Week 10. A period of increased run-off causes a small rise in TSS over the previous week's levels at Site 1. This is accompanied by TSS concentrations in silt pond effluent of between 100 and 350 mg/l, the latter value referring to Pond 2 (Figure 6). The difference between the inflowing and outflowing TSS concentrations at Pond 2 indicates that considerable silt is removed from the pond itself. At other times (Week 7) a lower incoming silt concentration (which is assumed to reflect lower run-off) does not affect the TSS levels at the outlet of Pond 2. A number of variables could be responsible for this difference in pond efficiency. These include flow rates and discharge, pond depth (see REYNOLDS *et al.*, 1990), and pond configuration. The orientation of outlet in relation to inflow canal and stream channel could have a marked effect on the direction and velocity of water currents passing through the pond. The ponds have already been described. The manner in which effluent enters the river (flowing down over rocks at Pond 2; effluent flowing overland to the river) is the most obvious difference which may contribute to their different retention efficiencies, although factors like overall depth and current flow were not measured.

The hypothesis that a difference in sediment retention efficiency between ponds accounts for the lower TSS levels at Site B than at Site D does not explain the high silt concentrations at Site C, which are similar to those downstream and yet not affected by Pond 2.

Particle size of the sediment in suspension will have a direct bearing on the time it is maintained in suspension (WILLIAMS & MOORE, 1986). This in fact may be the key difference in the situation obtaining at Site B relative to those below. Sites C, D and E are situated 1km below Site B, where the gradient decreases 2 m/km over this distance (Figure 1; Table 2). The soil of the surrounding agricultural land appears to have a much higher clay content than do the soils further upstream, and this may explain the generally higher levels of TSS at these sites. If this explanation is accepted, it implies that, although the silt ponds and connected canal system may contribute to suspended sediment loads in the Lourens River, all overland runoff entering this stretch of river carries an increased load in comparison with upstream areas. The activities that would be implicated in the high silt loads of overland flow in this vicinity are excavations on the hillside above the dam (Figure 1) for the construction of a three-storey building, two storeys of which are to be beneath ground level, as well as ploughing of land to establish orchard.

Invertebrates and the effects of sediment

There were no significant differences in respect of numbers of individuals or number of taxa, between any two sites along the river, or in the changes at any one site from July to August, despite the appearance of substantial changes in the graphical representation of data (Figure 7, Site B; Figure 8, Sites A and B). This gives support to the opinion that simple abundance measures are not enough to determine the effects of pollution on invertebrate communities, not only because they do not take account of the types of species present, but also because the highly contagious distribution of invertebrates requires the use of non-parametric statistics for analysis, and these are not stringent enough to detect changes that may not be extreme, but which may nevertheless indicate pollution stress.

Classification techniques indicate three groups at 70% similarity level (Figure 10), through the cluster dendrogram. There is a closer relationship between Site C and the group of Sites D and E. The two-dimensional ordination from multidimensional scaling confirms that Site C (Group 2, Figure 12b) is less closely related to Sites A and B (Group 1) than to Sites D and E (Group 3). KING (1981) reports that each method of summarising cluster analysis reported above distorts the relationship between samples to some extent, but that together they provide strong evidence of the similarity of groupings of samples.

Dissimilarity may be interpreted as a measure, in ecological terms, of how different groups of samples (or sites in this case) are. However, the identity of taxa that contribute to differences, and knowledge of the biology of such taxa is essential to the correct interpretation of data, particularly in relation to identifying directional change. The most important aspect of the Simper analysis is that it provides an interpretation of cluster analysis which integrates qualitative data into a quantitative measure. With other measures the examination of the specific taxa or species that occur at different sites is unrelated to the quantitative measure that forms the basis of categorising the status of a site (RABENI & GIBBS, 1980).

This does not imply that an examination of all components of the benthic fauna is not essential to describe in detail the changes taking place in a system, and for this reason the information in Tables 11 and 12 is discussed together.

The key taxa contributing to the difference between Groups 1 and 2 are Chironomidae sp.E and sp.CC., and Coleopterans: Elmidae sp.A and Helodidae sp.A. Helodidae sp.A and Chironomidae sp.E (a member of the Tanypodinae) are both upper river community taxa (HARRISON & ELSWORTH, 1958), which are not found in Group 2.

Group 2 and Group 3 are dissimilar, in part because each one does or does not contain Plecoptera sp.A or Chironomidae sp.G (a member of the Orthocladiinae) — both mountain stream or upper river taxa (HARRISON & ELSWORTH, 1958; KING, 1982; J. King, pers. comm.)

All but one of the taxa contributing to dissimilarity between Groups 1 and 3 belong to the mountain stream community (Table 11, 12). In Group 3 these taxa are either not present or are drastically reduced in number.

Thus a definite trend can be observed in the faunal community down the length of the river. It was established that this reach formed part of the same zone during summer. In winter the zonation pattern of rivers receiving annual seasonal run-off tends to become far less distinct (J. King, pers. comm.), because high discharge, floods and turbulent conditions tend to induce a degree of longitudinal homogeneity in chemical parameters and hence faunal communities. It would follow from this therefore that the same fauna should be present at all sites sampled in this zone in winter.

In fact, quite a different picture emerges from this study. Not only is there a division of the zone into three "sub-zones", but the type of fauna that becomes progressively reduced downstream is particularly the mountain stream / upper-river fauna, which might be expected to be present throughout this reach. In 1986, before the start of the development of Vergelegen, an examination of the benthos at Fleur du Cap (which was situated at the same site as Site D) found "both mountain stream elements as well as foothill elements" (BEAUMONT *et al.*, 1986), recording, *inter alia, Lestagella pencillata, Ephemerellina barnadi* and *Cheumatopsyche sp.* (mountain stream), and *Baetis harrisoni, Castanophlebia calida* and *Aeschna sp.*. In the present study, five of these taxa were missing entirely and the other two occurred in only small numbers at this site.

Of all the taxa recorded in the August survey, fifteen were characteristic of a combination of mountain stream and upper river fauna (J. King, pers. comm.; KING, 1982; HARRISON & ELSWORTH, 1958). Eight of these were unique to Group 1 and a further three restricted to Groups 1 and 2 (Table 12). The mayfly families, Ephemerellidae and Leptophlebiidae virtually disappeared from sites downstream of Site B. Even the adaptable *Castanophlebia calida* (HARRISON & ELSWORTH, 1958) was decimated in numbers between Group 1 and Group 3, whilst *Lithogloea harrisoni* was similarly reduced from being the most commonly occurring mayfly to occurrence as single individuals in a sample.

Baetids are frequently described as adaptable (BEAUMONT *et al*, 1986; HARRISON & ELSWORTH, 1958) and were present in the lower stretches of the study site, although again in reduced numbers. An exception to this was *Acentrella spp.*, which is the only true mountain stream representative of the family Baetidae in these samples (J. King, pers.comm.). This species was once again restricted to Groups 1 and 2.

Plecoptera were well represented at all sites in this study, with two species occurring in Group 3 (Sites D and E). Plecoptera in Cape streams are usually rare (KING, 1981), but in this study occurred in winter densities at Site D of up to 30 m^{-2}. PECKARSKY (1985) found that predaceous stoneflies recovered from experimental cages that had been silted up were still alive, whilst prey communities of Plecoptera, and in particular Ephemeroptera, were uniformly reduced. Plecoptera in Cape streams are not predaceous; nevertheless the effect of high TSS concentrations, whilst affecting other taxa, did not appear to reduce the presence of these species. This order is often the first to disappear with the changes characteristic of the transition from upper river to lower reaches, such as increases in temperature, or with the presence of organic pollution (J. King, pers. comm.). Their presence throughout, from Site A to Site E, is further evidence that the faunal changes taking place in the study reach are not due to any natural zonation patterns.

TAYLOR & ROFF (1986) found that during "recovery" from the effects of road construction, *Baetis spp.* and *Ephemerella* increased in numbers, as did *Cheumatopsyche sp.* (Hydropsychidae), the latter by some 1600%. CHISHOLM & DOWNS (1978) and HYNES (1972) observed an early dominance of Ephemeroptera in a rehabilitating stream, followed by caddisfly larvae, thus supporting the general findings of TAYLOR & ROFF (1986). BARTON (1977) observed a decrease in net-spinning and an increase in case-building caddis fly larvae. The present study found an attenuation in numbers of four genera of Trichoptera, including both types identified by BARTON (1977). GRAHAM (1988), however, suggested that silt could decrease the quality of epilithic periphyton, and hence the food resource for scrapers and grazers. This could contribute to the reduced distribution of Leptoceridae, and Hydroptilidae in this study.

Recovery by recolonisation, most often by drift, appears to occur rapidly after perturbation is ended, according to a number of researches into the short-term effects of mostly highway construction on roads (BARTON, 1977; BARTON *et al.*, 1972; GAMMON, 1970), with complete recovery being achieved after a period of between 87 days and one year, although "recovery" has been defined as re-establishment of diversity in many of these studies. Longer term studies have recorded ongoing fluctuations in community structure for up to five years, and permanent alteration of faunal composition (TAYLOR & ROFF, 1986).

The present study thus reveals a dramatic decline in particular components of the faunal community in the Lourens River, which correlate to the occurrence of higher TSS concentrations, below areas of intense farm development, than observed in the natural stream above. The importance of these changes in fauna is that they indicate the disappearance of most of the mountain stream and upper river taxa, which are generally considered to be less tolerant of stream alteration and pollution. There is no recovery of this component of the fauna one kilometre downstream of the major source of TSS loads (Pond 2), although overland run-off from the farm appears to be an important contributor to the TSS increases, and may thus prohibit recovery at Site E.

Finally, although natural drift has been mooted as the process whereby recovery of disturbed reaches may occur, this process assumes that the cause of catastrophic drift or other faunal responses to pollution will have ended. In the presence of continuing silt loads, therefore, recolonisation of the affected areas of the Lourens River by macroinvertebrates may not be possible.

CONCLUSION

1. There is strong evidence from the changes that occur in the Lourens River study area that suspended solids enter the river in run-off in concentrations an order of magnitude higher than would naturally occur. This is a recent development — the benthic fauna in 1986 showed no signs of pollution stress — and is probably the result of drainage and intensified development, rather than a natural process.

2. Whilst the establishment of silt ponds to remove some of the suspensoids in drainage water has met with some success (Pond 1), the silt pond created in the vicinity of the highest sediment-laden run-off is not an effective barrier against detrimental effects of the above activity.

3. The changes taking place in the river fauna show a trend toward progressive elimination of mountain stream and upper river fauna. These changes are a clear sign of pollution stress, which is extreme in the case of most mayfly species. A notable exception is the continued presence of plecopterans, suggesting that,

although this group is sensitive to temperature and organic pollution, the levels of suspended solids do not adversely affect its survival.

4. It appears as if recovery from environmental stress may include fluctuations in community structure, which would produce different indications of the "health" of the system, depending on the time of sampling (in relation to these fluctuations). Long term effects of perturbation may be severely underestimated if the test for rehabilitation of a river community is measured solely by an index of diversity, and ignores changes in species composition.

5. The dynamics of particle transport and settlement in flowing waters are not extensively described in the literature (REYNOLDS *et al.*, 1990). Further study of these dynamics may assist in the development of management strategies.

6. Deterioration of the river environment of the Lourens River, as a result of TSS loads may be restricted to a small section of the winter upper river zone. Nevertheless, it could be avoided by improving the functioning of the silt ponds, and by more sensitive development practises. For example, the initiation of excavation and building activities in late autumn is certain to contribute to soil erosion and increased overland run-off, carrying with it large loads of sediment. Finally, there does not appear to be a systematic programme to stabilise the canal banks, and this could be addressed in order to reduce erosion.

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