ROAD SALT CONTAMINATION OF GROUNDWATER IN A MAJOR METROPOLITAN AREA AND DEVELOPMENT OF A BIOLOGICAL INDEX TO MONITOR ITS IMPACT

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Abstract—A survey of 23 springs in the Greater Toronto Area (GTA) of southern Ontario recorded chloride contamination levels, resulting from the winter application of road de-icing salt, ranging from <2 to >1200 mg l⁻¹. Chloride level measured in spring water was far more seasonally stable than that measured in surface (river) water, and thus the spatial pattern of Cl contamination indicated by the former was judged to be more reliable. Chloride contamination of groundwater in the GTA was strongly related to urbanisation, and at the four most affected springs increases of between 21 and 34% were detected over the period November 1996 to November 1997. The response of macroinvertebrates living in these springs to increasing salinity was examined with the aim of creating a biological index of contamination: the Chloride Contamination Index (CCI). A power function yielded a significant correlation between this index and the mean Cl concentration measured at each spring. Taxa were able to be categorised as either "tolerant" or "non-tolerant" of high Cl although none was unique to either end of the scale. However, from both field observations and salinity tolerance trials in the laboratory, the amphipod Gammarus pseudolimnaeus was found to be associated with source aquifers only mildly contaminated with Cl. Absence of this species from a spring, particularly if nymphs of the stonefly Nemoura trispinosa are present should indicate moderate to high contamination. © 1999 Elsevier Science Ltd. All rights reserved

Key words—road salt, groundwater, urbanisation, biological index, chloride, contamination

INTRODUCTION

Degradation of groundwater quality as a result of human activities is a major contemporary concern (Pearse et al., 1985; Rudolph et al., 1992; Nolan et al., 1997). In Canada, point and non-point sources of industrial, agricultural and household wastes jeopardize the groundwater supplies used for domestic consumption by over 82% of the rural population and 26% of the population as a whole (Hess, 1984; Pupp, 1985). Unabated due to weak protective legislation, these contaminants are creating significant problems in our groundwater resources for generations ahead (Cherry, 1987). Groundwater contamination poses a unique, two part challenge for environmental managers: (i) to predict the likely sub-surface behaviour of contaminants; and (ii) to develop courses of remedial action and methods of aquifer management that can alleviate the problem. A critical step towards these goals is the monitoring and assessment of changes in groundwater quality. Extensive work has been done on surface water monitoring. Field sampling protocols have been improved considerably in the last two decades and sensitive analytical methods are now available for many pollutants. Technically, it is now possible to produce exhaustive organic, inorganic, isotopic assessment of discrete volumes of water. However, for groundwater, as with surface water, such a chemical analysis-based approach is subject to two major disadvantages. Firstly, the overall toxicity of contaminants, particularly when several co-occur, is difficult to determine. Secondly, unless a longterm sampling programme is in effect, the data collected can seldom reflect changes in water quality over time. Moreover, it can be difficult and expensive to monitor groundwater samples routinely, especially where drilling is involved.

Benthic community-based biological methods as an essential component of integrative monitoring of surface waters have been well developed and increasingly adopted by regulation agencies (Hellawell, 1986; Plafkin et al., 1989; Karr, 1991; Rosenberg and Resh, 1993). The advantages include: (i) benthic communities are sensitive to a
wide range of water quality degradation; (ii) they can indicate the overall toxicity (cumulative and synergistic) of contaminants; and (iii) as they live permanently submerged they can reflect changes in water quality over time. Biological monitoring therefore could and should be applied to groundwater contamination. Considerable work has been done on the ecotoxicology of groundwater and the ecology of groundwater faunas (see Notenboom et al., 1994, for a review), and strong relationships have been demonstrated between contaminants and groundwater invertebrate community structure (Sinton, 1984; Malard et al., 1996). To overcome difficulties in accessing groundwater and its fauna, Danks and Williams (1991) proposed that the invertebrate faunas of freshwater springs, these being natural outlets of groundwater, could be used for monitoring and assessing groundwater quality. The very stable physical and chemical conditions found in springs (Forester, 1991) make it possible to investigate the relationships between spring faunas and groundwater quality more rigorously than in surface waters, for example. Further, the accumulation of aquatic plant and invertebrate fragments in spring sediments can provide a historical record of water quality spanning tens if not hundreds of years (Williams, 1989). The potential of spring communities in monitoring groundwater is being increasingly recognised by ecologists (Kamp, 1995; Notenboom et al., 1997; Sarkka et al., 1997).

In a preliminary study, we analysed the macroinvertebrate communities of a series of springs in south-eastern Ontario in relation to water quality (Williams et al., 1997). The data showed some close linkages between certain species/species assemblages and major water quality variables. The present study focuses on one of the most invasive chemicals that pollutes groundwater in snowbelt countries, road de-icing salt (NaCl). Salt is widely used in many northern countries and the problem is acute and increasing around cities because of their large road surface area. In the Greater Toronto Area (GTA), for example, more than 100,000 tonnes of salt are currently applied each winter—a rate of approximately 200 g for every square metre of land (Howard and Beck, 1993). In one GTA catchment—the Duffin Creek-Rouge River basin—45% of the salt applied is removed by overland flow, with the remaining 55% entering temporary storage in shallow groundwaters (Howard and Haynes, 1993). In the past few years, Cl− concentrations > 1000 mg l−1 have been recorded frequently in Highland Creek, one of the major rivers in the basin (unpublished data, Ontario Ministry of Environment and Energy). This is more than 4 times the standard for drinking water (250 mg l−1). Even higher Cl− levels (10,000 mg l−1) can be found in shallow sub-surface waters in other parts of the GTA (Taylor, 1992). Elevated Cl− levels in algae are known to interfere with photosynthesis (Boyle, 1984), whereas in higher organisms they result in potentially fatal metabolic acidosis and osmotic stress, and in behavioural changes (Martem’yanov, 1989; Tytler and Ireland, 1995).

Apart from documenting the levels of contamination of groundwater by salt in the GTA, through monitoring spring water, a second purpose of this study was to attempt to develop a biotic index for assessing and monitoring this contamination. The approach to the latter was twofold: through a field survey of the graded responses of natural spring invertebrate communities to salt contamination, and through laboratory bioassays of salt tolerance by possible indicator species.

**STUDY AREA AND SPRINGS**

A temporal-spatial analysis of salt level distribution within a given area was fundamental to the study in order to both confirm the source of salt contamination and to compare the stability of spring and river water chemistry. For this we used environmental data, provided by the Ontario Ministry of Environment and Energy (O.M.E.E.), on several rivers/streams in the study area—comprising several neighbouring catchments (Highland Creek, River Rouge, and Duffin Creek) within the GTA (Fig. 1). The northern boundary of the selected area lay in the Oak Ridges Moraine, while the southern boundary was the northern shore of Lake Ontario in Suburban Toronto. From these data, we analysed the changes in Cl levels over several years (1990–1994) at many sampling sites.

Within the area, the bedrock geology consists of Upper Ordovician shales through which a relict (buried) drainage system runs. The unconsolidated glacial/interglacial sediments that fill this channel support many aquifer systems (Eyles et al., 1985), which provide water to industrial and municipal consumers and to local farms. The hydraulic characteristics and hydrochemistry of 14 major aquifers have been studied extensively (Howard and Beck, 1986). Growth in the GTA in recent years has been unprecedented and, as a consequence of increased human activities, the groundwater quality has deteriorated due to a variety of contaminants (Howard et al., 1993). There is a clear urban–rural transition from Highland Creek in the west to Duffin Creek in the east. The Highland Creek catchment is largely within Metropolitan Toronto and, with an area of 109 km², it is the most heavily urbanised (> 80%; Howard et al., 1993). Its industrial and dense residential areas are supplied by a dense network of roadways and multi-laned highways. The Rouge River catchment (293 km²) is less densely populated, particularly in its northern part, where the river arises from the Oak Ridges Moraine and runs through woodlands and farmlands for much of its length before entering an urbanised southern part. Duffin Creek also orig-
inates in the Oak Ridges Moraine, and its catchment of 333 km² largely consists of forest, woodland/grassland and farmland except for a southern, urbanised section. Further details of these catchments can be found in Sibul et al. (1977), Howard et al. (1993) and Taylor (1992).

Within this study area, we located over 30 springs; however, some proved to be seasonal and were later excluded from the analysis. Of the remaining 23 springs, Springs 1–16 were sampled in order to examine the relationships between spring macroinvertebrate communities and salt contamination, and to develop the biotic index. Springs 17–23 were used, subsequently, to validate the effectiveness of the index. The distribution of these springs is shown in Fig. 1 and brief descriptions of each are given in Table 1.

**SAMPLING METHODS AND DATA PROCESSING**

In order to assess the impact of road salting on surface and groundwater quality, surface water Cl levels in the study area (from the O.M.E.E. dataset) were compared with the spatial-temporal distribution of Cl levels recorded at the spring sites. To establish the latter, water samples were collected monthly at each of Springs 1–16, at a point not more than 2 m downstream of the source, from November 1996 to November 1997. Chloride concentrations were assessed using the Hach Kit procedure.
Spring Location Type Substrate and Vegetation Land-use
1 44°00'N-79°05'W Rheocrene Sand + mud; watercress Woodland
2 43°59’N-79°17’W Limnocrene Mud + sand; algal mat Meadow
3 43°59’N-79°17’W Limnocrene Silt; watercress Meadow
4 43°59’N-79°17’W Helocrene Silt + fallen leaves Woodland/meadow
5 43°59’N-79°18’W Rheocrene Silt Cedar woodland
6 43°59’N-79°18’W Limnocrene Silt + wood debris; watercress Cedar woodland
7 43°59’N-79°18’W Limnocrene Silt + wood debris Cedar woodland
8 43°50’N-79°00’W Rheocrene Sand + silt + fallen leaves Forest
9 43°50’N-79°00’W Rheocrene Sand + silt + fallen leaves Forest
10 43°50’N-79°00’W Rheocrene Sand + silt + cedar leaves Cedar woodland
11 43°51’N-79°00’W Rheocrene Gravel + sand; moss/algae Grass/woodland
12 43°51’N-79°11’W Rheocrene Silt Meadow
13 43°47’N-79°12’W Helocrene Silt + sand By road and bridge
14 43°45’N-79°15’W Rheocrene Sand + gravel + silt; watercress Woodland
15 43°45’N-79°15’W Rheocrene Sand + silt; moss Woodland
16 43°45’N-79°12’W Limnocrene Sand + gravel Woodland by road
17 44°00’N-79°04’W Rheocrene Sand + silt Forest
18 44°00’N-79°04’W Limnocrene Sand + silt Forest
19 44°00’N-79°04’W Limnocrene Sand; moss Forest
20 43°43’N-79°13’W Rheocrene Sand + silt Woodland; urban
21 43°43’N-79°13’W Rheocrene Sand + silt Woodland; urban
22 43°43’N-79°13’W Rheocrene Sand + silt Woodland; urban
23 43°43’N-79°14’W Rheocrene Silt + mud Woodland; urban

(Hach, 1990). Water temperature and conductivity were measured also at these times, using field meters. Benthic samples were taken in November 1996 using a 250 μm mesh hand net to collect invertebrates dislodged from a 20 x 20 cm area of spring bed (to a substrate depth of 5 cm). Two such samples were taken from each spring so as to include major microhabitat types. The two samples were combined and preserved in a 10% solution of formalin and Rose Bengal stain. Although the absolute sample size was small (0.12 m²), in terms of the total habitat areas of these small springs it represents a much larger percentage than is typically sampled in running water studies.

Chloride measurements and benthic samples were taken at the start of each trial, 10 similar-sized individuals of one species were introduced into each bowl. Survival and condition were assessed after 24, 48, 72 and 96 h at three levels: dead, stressed and healthy. Stress was defined by two or more of the following behaviours (depending on species): cessation of feeding; abandonment of case or failure to construct a feeding net (caddisflies); change of colour (amphipods). To assess the chronic effects of Cl contamination, we used two species known to withstand captivity well, Gammarus pseudolimnaeus and Physa sp., at two treatment levels, 1000 and 2000 mg l⁻¹, together with controls. The tests lasted for two months during which time the animals were fed pelleted fish food.

RESULTS

Chloride contamination in groundwater and surface water

Variation in Cl concentrations in Highland Creek, the Rouge River and Duffin Creek over 4
years are shown in Fig. 2. Chloride levels were generally highest in Highland Creek and there was significant variation over time. Typically, Cl reached peaks in January/February, with a maximum of 1390 mg l\(^{-1}\). The mean level was 316 mg l\(^{-1}\) with a minimum of 22 mg l\(^{-1}\). A similar seasonal pattern was observed in the Rouge River, but the peaks were much lower (maximum of 970 mg l\(^{-1}\); mean = 162 mg l\(^{-1}\)). Duffin Creek showed the lowest contamination (mean = 107 mg l\(^{-1}\)), without obvious seasonal peaks. These data indicate that Cl levels throughout the study area are strongly related to urbanisation, with Cl increasing as landuse moves from rural to urban. Moreover, the strong seasonal pattern correlates high Cl levels with peak snowfall and concomitant road-salting.

Cl concentrations at Springs 1–16 varied substantially over space (Fig. 3). The lowest level was recorded at Spring 1 in the Glen Majors Conservation Area (mean = 2.1 mg l\(^{-1}\)). Higher levels occurred at some of the rural springs (2–4, 9, 10; mean Cl = 100 mg l\(^{-1}\)). Comparison of the Cl levels measured in the four most contaminated springs (13–16) showed increases ranging between 21 and 34% over the period November 1996 to November 1997 (Fig. 4). The highest Cl level was recorded at urban Spring 13, which is adjacent to a bridge and highway (maximum = 1345 mg l\(^{-1}\); mean = 1092 mg l\(^{-1}\)). Other high Cl springs were similarly located close to major urban salting sites (e.g., 14–16). Overall, as for surface waters, the Cl levels found in springs waters, were closely correlated with degree of urbanisation. A major difference, however, was the far greater seasonal stability of the Cl level in the springs (Fig. 3). This implies that the spatial pattern of Cl contamination indicated by spring water is more reliable and accurate than that measured in river water. Further, for monitoring groundwater contamination, it is much easier to sample springs than to collect groundwater directly (which typically involves drilling).

Community analysis

The species composition and relative abundance of macroinvertebrates found in Springs 1–16 are shown in Table 2. A total of 34 taxa was recorded, comprising 20 types of insect, 5 cladocerans, 3 molluscs and 6 others. Chironomids were found in almost all of the springs, and oligochaetes, tipulid larvae (Diptera), ostracods, Nemoura trispinosa (stonefly) and Lepidostoma sp. (caddisfly) occurred in over half. Interestingly, about 50% of the taxa were collected only at one site. This indicates, as in many other habitat types, that rare species constitute the majority of spring benthos biodiversity. The number of taxa in each spring ranged from 5 to 11, with a mean of 8.

The CCA ordination of the 34 taxa against Cl levels is presented in Fig. 5. The first two axes explained 27% of the total variance. In the first two dimensional space, the taxa were spread widely, with the species usually found in low Cl springs occurring to the left and species largely associated with high Cl springs appearing to the right. The cluster analysis based on species scores on the first CCA axis using Average Euclidean Distance and

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![Fig. 2. Changes in chloride concentration in surface waters within the GTA over 4 years (1990–1993).](image-url)
Ward Linkage (Fig. 6) shows two groups. The first consists of 13 taxa (including oligochaetes, *Crangonyx* sp., copepods, the snails *Lymnaea* sp. and *Physa* sp., nematodes, ceratopogonid and tipulid dipterans and *Stratiomys* sp., and several other rare taxa). The second comprises the remaining 21
taxa—most of them largely restricted to the low Cl springs (e.g., *Gammarus pseudolimnaeus*, the beetle *Hydroporus* sp., the caddisfly *Molanna* sp., and the stoneflies *Nemoura trispinosa* and *Soyedina* sp.). However, some members of the latter group occurred at a range of Cl levels (e.g., chironomids, ostracods, and the clam *Pisidium* sp.).

The biotic index

A tolerance value for each taxon was assigned according to which cluster it belonged to in Fig. 6: all taxa in Group 1 were given a score of 5 while those in Group 2 were scored 10. Thus the higher the value, the lower the tolerance to Cl. The Chloride Contamination Index for Cl (CCI) was calculated as follows:

$$\text{CCI} = \left( \sum X_i \times T_i \right) / N$$

where:
- $X_i = 1$ if taxon $i$ was present, and $X_i = 0$ if absent
- $T_i$ = tolerance value of taxon $i$
- $N$ = the total number of taxa at the spring site

CCI reaches the minimum of 5 when all taxa at a spring fall within the low-score group, indicating high Cl contamination, and achieves the maximum of 10, when all taxa belong to the high-score group, indicating no Cl contamination.

This new index was applied to Springs 1–16 and the calculated CCIs were plotted against the average Cl concentrations measured at each spring (Fig. 7). Several mathematical relationships were fitted to the curve, including linear, exponential and power functions, with the latter yielding the highest regression coefficient ($r = 0.804$; $P < 0.05$). The majority of the spring sites, 1–16, fell around the fitted curve. Spring 16 proved to be an exception as it achieved a CCI value as low as at Spring 13, yet its Cl level was much lower.

Field assessment

In order to validate the effectiveness of the CCI, the relationship between the CCI value and Cl concentration was investigated using Springs 17–23 (Fig. 7). Springs 17–19, located in the rural, Oak Ridges Moraine (where Cl levels are very low; 15–20 mg l$^{-1}$) achieved high CCI values (9.1–9.4). In contrast, Springs 20–23 which are located within Metropolitan Toronto, and have much higher Cl levels (220–455 mg l$^{-1}$), achieved lower CCI values (6.7–7.5). These findings indicate that the CCI works well in predicting Cl contamination levels of spring water within the study area.

Salinity tolerance tests

All species tested at 3000 mg l$^{-1}$ Cl level survived
for 96 h without obvious stress. At 4500 mg l\(^{-1}\) (Table 3), Physa sp., a “tolerant” species from the field survey, became stressed (it stopping feeding and moving) although it recovered a few hours after being transferred to fresh water. Another member of the tolerant group, Crangonyx sp., survived well for the first 24 h, but thereafter turned slightly pink in colour and exhibited difficulty in swimming; after 96 h, nine of ten individuals were stressed. At the 4500 mg l\(^{-1}\) level, the performance of “non-tolerant” species from the field survey varied considerably. Gammarus pseudolimnaeus began dying within 24 h, and was the most sensitive. However, Lepidostoma sp., and Nemoura trispinosa survived better, with nearly all individuals appearing normal at the end of 96 h. The survival of Parapsyche sp. was more difficult to assess as some cannibalism occurred in the bowls. However, survival in the test and control groups was similar, particularly after 72 and 96 h.

Salinity tolerance at 6000 mg l\(^{-1}\) was similar to that seen at 4500 mg l\(^{-1}\) (Table 3), but the effects were more pronounced. Notably, most of the Physa sp. died after 72 h, Crangonyx sp. began dying after 24 h, and all of the G. pseudolimnaeus were dead after 96 h. Lepidostoma sp. survived well for 48 h, but 50% were dead after 96 h. Nemoura trispinosa exhibited the greatest survival (70% after 96 h).

Again, allowing for cannibalism, there was little difference in the survival of Parapsyche sp. between the tests and the controls.

In the longterm trials (Fig. 8), there were no differences between survival in the tests (1000 and 2000 mg l\(^{-1}\) Cl) or controls for either Physa sp. or G. pseudolimnaeus, although 100% of individuals survived in the former compared with ~80% survival in the latter. Although not measured quantitatively, there was a visible size increase in the snails over the 2-month trial period. At the end of this time, the control group amphipods began mating but this did not occur in either of the Cl treatments.

**DISCUSSION**

The relationship between salt contamination and aquatic communities is poorly known (Crowther and Hynes, 1977; Dickman and Gochnauer, 1978). Yet Pupp (1985) and Cherry (1987) both have predicted that salting poses a major threat to freshwater environments in cold regions.

The results from the present spring survey and analysis of surface water data confirm that Cl contamination in the study area is related to urbanisation and, specifically, the use of road de-icing salt. Springs emerging from groundwater in the Oak...
Assessing road salt in groundwaters

Fig. 6. Cluster analysis of the species/taxa based on the scores generated on the first CCA axis, using Ward Linkage and Euclidean Distance.

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Table 3. The effects of high Cl levels on the survival of 6 spring macroinvertebrate species in the laboratory at 7°C (H=healthy, S=stressed, D=dead No. out of 10 individuals)

<table>
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Assessing road salt in groundwaters
Ridges Moraine provide a baseline for Cl level, which is below 20 mg l\(^{-1}\). Chloride levels of groundwater in the urban areas are currently 10–60 times higher. This reflects the substantial influence that road-salting is having upon freshwater resources in Canada's largest metropolitan area.

![Correlation graph](image)

**Fig. 7.** Correlation between the Chloride Contamination Index (CCI) and chloride concentration as measured in 23 springs within the GTA.

![Survivorship graph](image)

**Fig. 8.** Survivorship of the snail *Physa* sp. and the amphipod *Gammarus pseudolimnaeus* at three chloride levels over 60 days.
Our analysis clearly shows that spring water provides a more stable indicator for determining Cl contamination. Routine sampling of selected springs throughout the GTA thus will provide a good record of the present extent of contamination together with any future increases. Such sampling may be done either by the more traditional chemical analysis of chloride ions, or by using the biological index developed here—the latter providing more toxicological information.

In the past, most biotic indices were created for monitoring and assessing organic pollution in surface waters (Hellawell, 1986; Metcalfe, 1989; Rosenberg and Resh, 1993), although several are now aimed at heavy metal contamination (Clements et al., 1992). The present index is unusual, in that it attempts to characterise an important, modern contaminant of groundwaters, one that is different from both organic and metal forms. Our study shows that from benthic samples taken at randomly selected springs, it is possible to distinguish groups of species that are associated more with high or low Cl levels. Within the range of Cl levels measured, however, we did not find taxa that were unique to either end of the scale (although none of the “non-tolerant” species was found in Spring 13, the most contaminated) and thus our field survey did not identify definitive “indicator species” per se. However, the laboratory assays did identify the amphipod *Gammarus pseudolimnaeus* (and also, to a lesser extent, the amphipod *Crangonyx sp.*) as being intolerant of both acute and chronic exposure to high Cl, and in having its reproductive behaviour disrupted by the latter. In GTA springs *G. pseudolimnaeus* never occurred as more than one or two individuals in the benthic samples from high Cl sites, yet it was abundant in samples from low Cl sites. In contrast, nymphs of the stonefly *Nemoura trispinosa* survived the salinity trials well and were found at high densities in contaminated springs. As a first biological indication, therefore, we suggest that spring samples containing high numbers of *G. pseudolimnaeus* should indicate source aquifers only mildly contaminated with Cl. Absence of this amphipod in a spring, particularly if nymphs of the stonefly *Nemoura trispinosa* are present should indicate moderate to high contamination. *Nemoura trispinosa* does not appear to be a sufficiently good indicator on its own as, in the field survey, it ranked with “non-tolerant” species—which underlines the importance of using both field and laboratory assessment protocols.

Greater precision in determining the actual Cl level would necessitate closer examination of the benthic samples and calculation of the CCI. The latter is a relatively simple procedure not requiring a high degree of technical training. Faunal differences between geographical regions would require that a new CCI be established, using the method developed here, for use outside southern Ontario.

The laboratory trials showed that Cl levels of 4500–6000 mg l⁻¹ were lethal, within 96 h, for most of the spring invertebrate species tested. Although we did not measure levels as high as these in the 31 springs sampled, levels as high as 10,000 mg l⁻¹ have been recorded in shallow subsurface waters at some urban sites within the GTA (Taylor, 1992). Further, the inter-year comparison of Cl levels at four of the most contaminated springs showed increases ranging from 21 to 34% between 1996 and 1997. If this is a true reflection of the accumulation of Cl in aquifers of the GTA (by comparison, the Cl level in a Norwegian lake increased by 200% from 1966 to 1991 as the result of road salting: Kjensmo, 1997), then the potential of spring invertibrates and the CCI to indicate progression of this contaminant will be confirmed quite soon.

**CONCLUSIONS**

1. Chloride contamination of groundwater, from road de-icing salt, in the Greater Toronto Area was strongly related to urbanisation, and at some sites increased between 21 and 34% from November 1996 to November 1997.

2. The spatial pattern of Cl contamination indicated by spring water proved to be more stable than that measured in surface (river) water.

3. CCA ordination and subsequent cluster analysis divided spring-dwelling invertebrates into being either “tolerant” or “non-tolerant” of high Cl, although none was unique to either end of the scale. A new biotic index of contamination—the Chloride Contamination Index (CCI), which summarises the Cl tolerance of a spring community—was significantly correlated with the Cl concentration measured at the springs.

4. The amphipod *Gammarus pseudolimnaeus* was associated with source aquifers only mildly contaminated with Cl. Absence of this species from a spring, particularly if nymphs of the stonefly *Nemoura trispinosa* are present should indicate moderate to high contamination.

5. Springs are recommended as easy access points for study of both chemical and biological aspects of aquifer contamination.

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**REFERENCES**


