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SEWAGE: TOWARDS REALISTIC ENVIRONMENTAL PROTECTION

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ABSTRACT

The disposal of sewage into aquatic systems is attracting increasing regulation at the European Community level. Initially, pollution control strategies focussed on the protection of rivers and inland waters from excessive oxygen demand loading. This has led to an increasing use of marine waters for the disposal of sewage effluents and sludges. The bacteriological hazards associated with these practices are now well understood and have led in turn to restrictions on such disposal activities. This problem is illustrated with data from Venice Lagoon showing extreme bacterial contamination. A less commonly appreciated problem relates to the chemical contamination of sewage from both household and industrial chemicals. Results from analyses of a variety of sewage effluents are presented in this paper and the implications of the inventory of chemicals found are discussed.

KEYWORDS

Sewage, organic contamination, metals, environmental effects.

INTRODUCTION

The deleterious effects of sewage discharges upon river quality have been extensively studied. Until 200 years ago, river deterioration in the UK was not serious due to the relatively small human population living in widely distanced communities. The human health aspects of introduced pathogenic organisms were, admittedly, a severe problem, although the bacteriological link had not been established (Mason 1991). With the industrial revolution there came a rapid growth in population and increasing urbanisation. Raw sewage was discharged in large quantities into watercourses leading to severe deterioration in water quality and frequent outbreaks of diseases such as cholera and typhoid.

In the UK, efforts to clean up the major rivers were given particular impetus by the nuisance caused by the highly polluted River Thames in the mid-1800s. The remedial measures taken since the 1950s (Wood 1982) have resulted in an intensively managed river system where oxygen levels in the tidal reaches are rigidly monitored, controlled and maintained (Ellis 1989). Treated effluents are discharged to the river while sludge is dumped in the outer estuary. This has resulted in a steady improvement in the biological quality of the river since 1960 (Andrews 1984). A similar management system using oxygen addition is being developed for the River Seine in France (Patel 1992) following severe fish kills due to oxygen depletion, caused in turn by storm sewage overflows. In both cases, the river itself is incorporated into the treatment process on the basis of its notional assimilative capacity.

Currently, in England and Wales it has been estimated (Lester 1990) that exclusive of cooling water, 23 million cubic metres of water are used daily. Domestic use accounts for 8 million cubic metres which together with approximately 6.8 million cubic metres of the water used by industry is discharged to sewer. Overall, the sewage produced by 44 million people is treated to varying degrees at the 5000 or so treatment works serving populations in excess of 10,000. Sewage from 6 million people is discharged untreated to the sea while between 1 and 2 million people are not connected to the sewerage system.

Despite the relatively high level of treatment of sewage in the UK, gross pollution of rivers is still a problem. Since 1980 the trend of improving river quality in England and Wales has been reversed and between 1985 and 1990 a decline in quality was reported for 3.6% of the river lengths surveyed (NRA 1991a). The most common cause cited by each of the ten administrative regions was the poor performance of sewage treatment works resulting in high BOD and ammonia loadings on the waters concerned. In short, the notional self-purification ability of the receiving water has been overwhelmed.

The high profile given to ameliorating gross pollution of this kind has, until recently, drawn attention away from other environmental aspects of sewage discharges. This paper considers two important aspects of sewage which are the subject of intense study. Both problem areas have arisen as a result of considering the natural environment simply as an extension of the processes taking place in treatment plants. They have long-term implications for the regulation of such discharges since it is obvious that they are founded on fundamental misconceptions of the processes occurring within and outside treatment works.

Microbiological quality problems are illustrated using results obtained from Venice Lagoon. These are discussed comparatively with the UK situation in the context of complying with EC legislation. The problem of chemical content due to domestic and industrial sources is considered in relation to qualitative analytical results obtained from chemical screening of sewage effluents and discussed in terms of the difficulties of adequate regulation.

METHODS AND MATERIALS

Microbiological Analyses Ten sampling points in Venice Lagoon were selected to include outlying channels and waters in the highly populated centre of Venice. The sampling points are shown in Figure 1. Sub-surface water samples (200ml) were collected in sterile containers and held on ice until processing. 1ml and 10ml volumes were filtered through Gelman 0.45μm filters and these were then incubated on pads soaked in lauryl sulphate broth for 16 hours. Duplicate plates were run at 37°C and 44°C using a PAQUALAB portable unit to determine total and faecal coliforms respectively. At the end of the incubation period, colonies having a persistent colouration on cooling were counted.

Chemical Screening of Sewage Effluents. 1 litre samples of sewage effluents were collected in prepared glass bottles and transported chilled at 4°C to the laboratory. A 100 ml subsample was taken for analysis of metal content. Samples were spiked by the addition of a standard of deuterated naphthalene to produce a final concentration of 200 μg l^{-1}. The effluents were then extracted into 10ml of hexane using a separatory funnel and 1μl of the extract was injected into a Hewlett-Packard 5890 Gas Chromatograph connected to a Hewlett-Packard 5970 Mass Selective Detector. The temperature was ramped from an initial 35°C at a rate of 7°C min^{-1} to a final temperature of 280°C. The column used was an Ultra 1 (cross linked methyl silicone gum phase), internal diameter 0.2 mm, stationary phase thickness 0.33 μm and 25m in length. The carrier gas was helium. Compound identification was by probability-based matching of mass fragmentograms with those held on the US National Bureau of Standards Library. Metal analyses were conducted using a Varian Liberty 100 ICPAES (inductively coupled plasma atomic emission spectrometer). The ICPAES was calibrated against commercially available ICP standards (MBH Analytical U.K.) and standard quality control checks were carried out on each batch of samples.

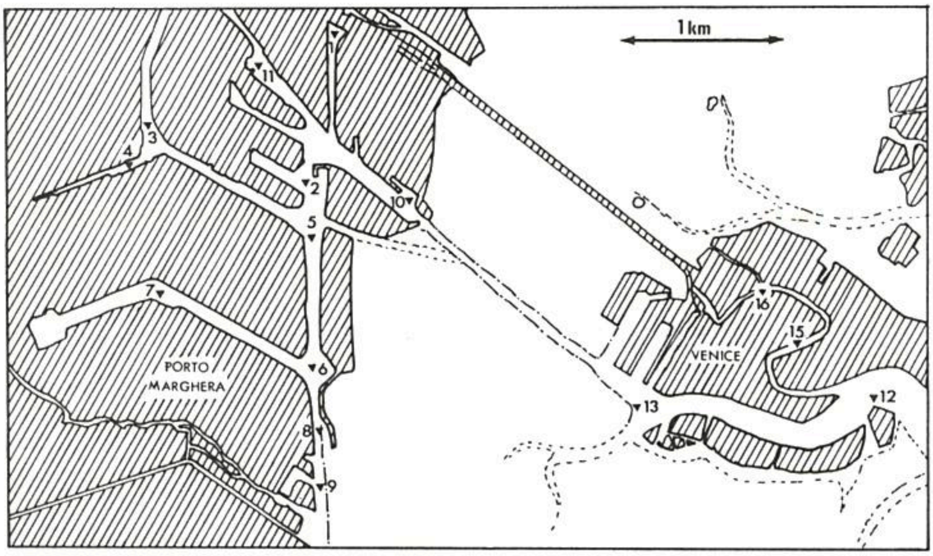

Figure 1: Map of Venice Lagoon showing sampling sites used for bacterial determinations

RESULTS

Microbiological analyses The results of microbiological analyses made on sub-surface water samples taken from Venice Lagoon are shown in Table 1 below.

Table 1: Total and Faecal coliforms determined in water samples from Venice Lagoon. Sample sites are shown in Figure 1.

Sample (SITE)	Total Coliforms /100ml	Faecal Coliforms /100ml
1(1)	3200	n/a
2(2)	700	100
3(3)	6400	930
4(4)	8300	180
5(5)	2500	220
6(6)	-	-
7(8)	1000	180
8(12)	70	10
9(15)	12300	10400
10(16)	5800	5700
EEC guideline*	500	100
EEC mandatory*	10000	2000

* Council directive 76/160/EEC December 1975

Chemical Analyses. The analytical traces obtained from the GC/MS screening of hexane extracts of the effluents are displayed in Figures 2a & 2b. Trace metal results from the sewage effluents determined using ICPAES are also shown on the individual traces together with the number of compounds isolated from each using GC-MS sample. The percentage of these which could be readily identified by comparison with the US National Bureau of Standards Spectral Library at the 50% and 90% level are indicated.

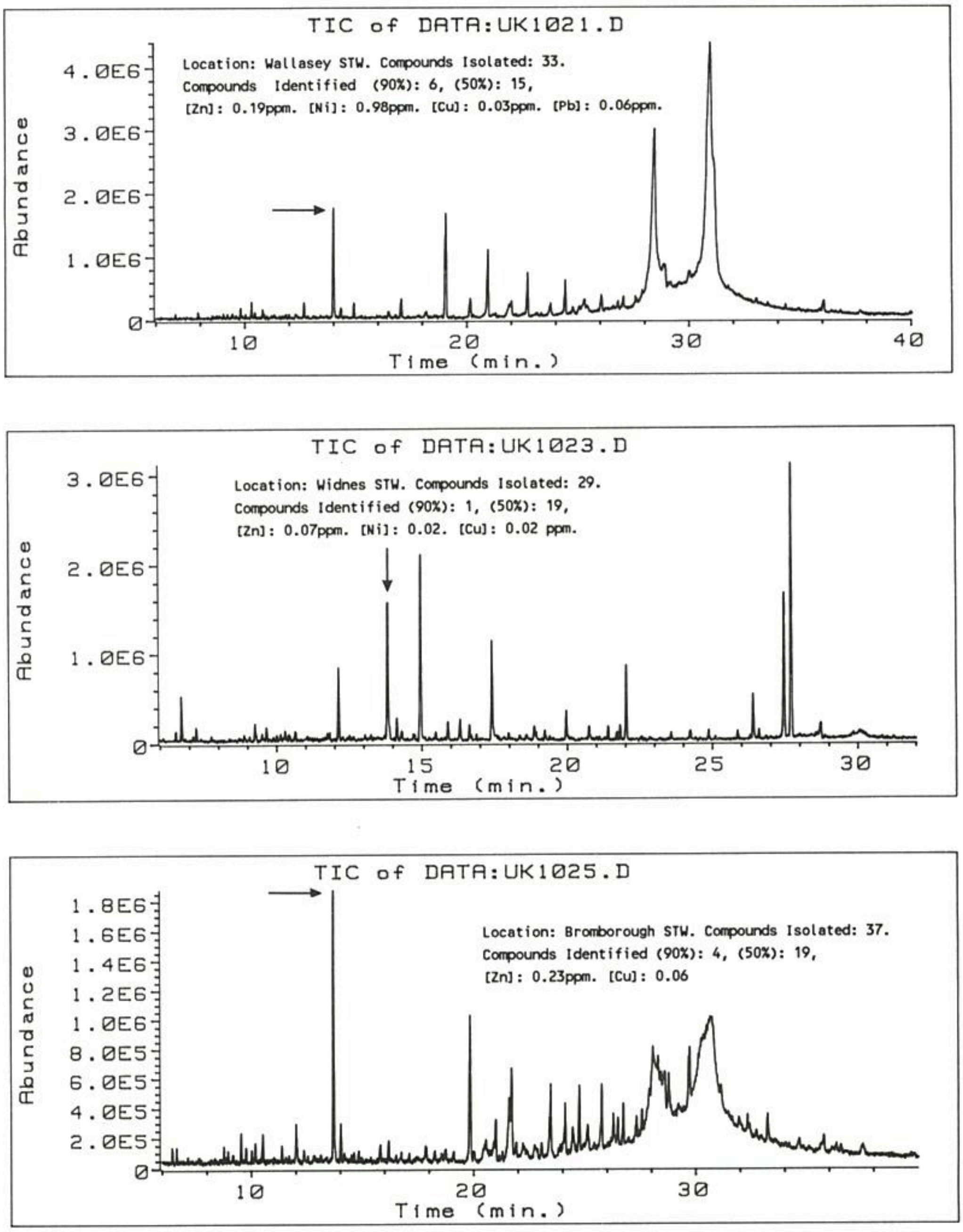

FIGURE 2a: Total Ion Chromatograms obtained from the analysis of sewage effluents from the Mersey area in the UK. The arrow identifies the peak due to the internal standard added at 200ppb. Numbers of isolated and identified compounds are given. Concentrations of metals were determined by ICPAES.

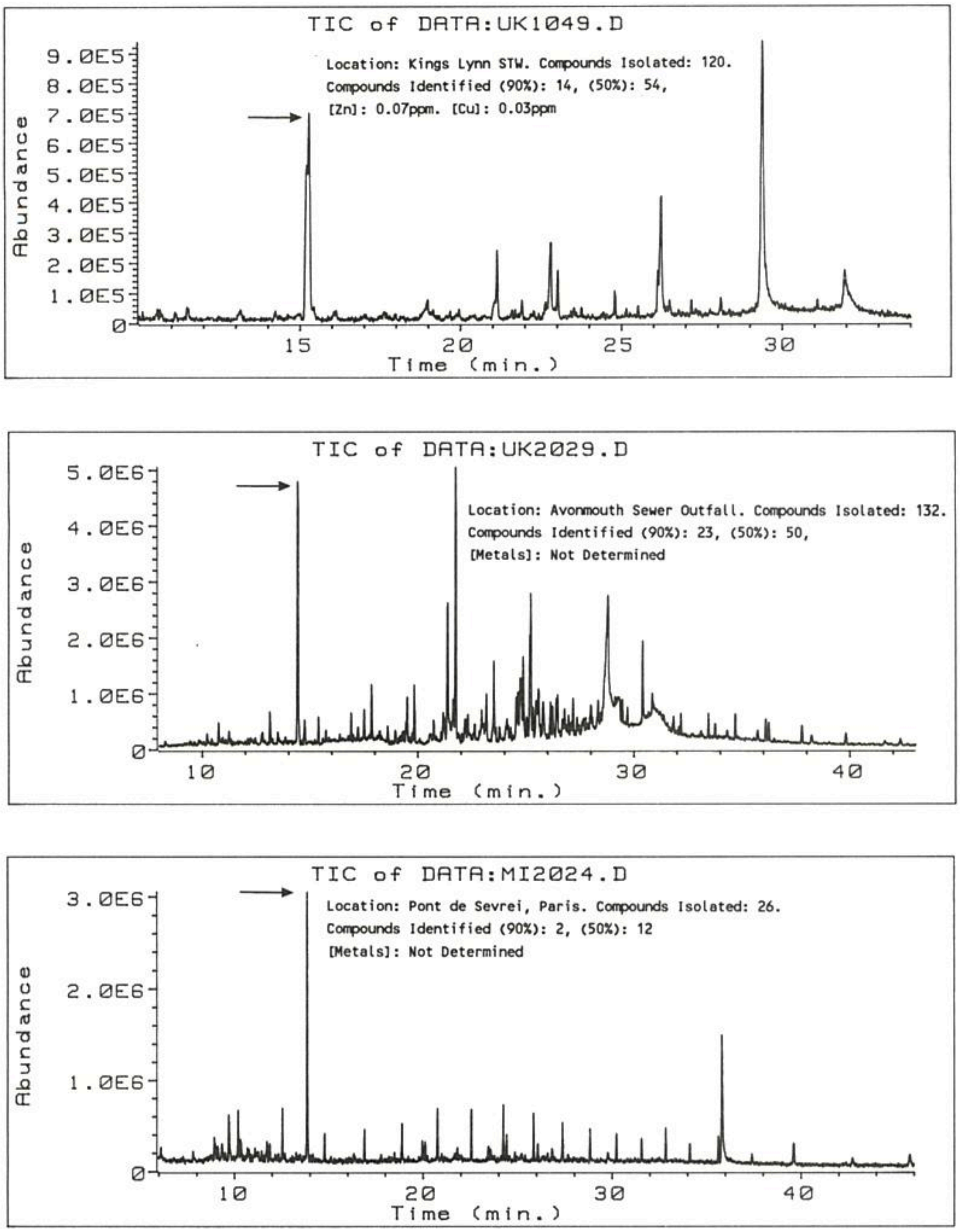

FIGURE 2b: Total ion chromatograms from the analysis of sewage effluents from the UK and France. The arrow indicates the internal standard peak. Numbers of isolated and identified compounds are shown. Metal levels were determined by ICPAES.

The baseline shift apparent in some of the traces at around 30 minutes is due to the presence of fatty acids from faecal material and the magnitude of the shift can serve as an approximate index of the relative proportions of grey, black and industrial waste waters entering the works.

Nonetheless, all the effluents from the Mersey area contained chemicals of probable industrial origin. Substituted benzenes were commonly found together with PAHs and industrial solvents. 2,4,6-trichlorophenol was present in one discharge. Most of these compounds appear to be present in part per billion quantities. The mass balance, however, is likely to be high due to high effluent flow rates. Sample UK1049 was found to contain trichloromethane together with three chlorinated benzenes. Chlorinated phenolics, phthalates, and PAHs were found in sample UK2029. The Parisian sample is of interest due to the low identification achieved despite the obvious complexity of the analytical trace.

DISCUSSION

Microbiological quality. The Lagoon of Venice has a surface area of some 549km^2. Lagoon waters are exchanged with the sea through three openings to the Adriatic Sea. Freshwater inflows with associated contaminant burdens enter the lagoon body through more than 20 points distributed on the inner lagoon border. Industrial and sewage effluents discharge directly into the lagoon. Due to its enclosed nature the lagoon is a low energy area with a mean depth of approximately 0.6m (Raccanelli *et al.* 1989). Venice is highly populated particularly during the summer months due to the influx of tourists and in recent years eutrophication has become an increasing problem in the lagoon (Pavoni *et al.* 1990).

The results indicate that Venice Lagoon is highly contaminated with sewage bacteria. At the time of sampling (March 1990) coliform levels were, with one exception, equal to or in excess of EEC guidelines for bathing water. The sample taken at site 15, in the centre of Venice, is above EEC mandatory levels for both total and faecal coliforms. These are defined in the EC Bathing Water Directive (EEC/76/160). Although actual bathing is infrequent in the lagoon itself, recreational beaches exist on the littoral strip on the Adriatic Sea.

The results are a clear illustration of the assertion of Anderman & Maritim (1986) that improper sewage treatment and disposal is currently a major factor threatening the health and comfort of individuals in areas where municipal sewerage systems are inadequate. In Venice there has been a historical exploitation of its unique positioning for direct disposal of sewage into the canals, and failure to develop a treatment infrastructure in the surrounding areas. Considerable reduction in the numbers of pathogenic organisms present in sewage can be achieved by full treatment, but nonetheless, even in waters receiving treated effluents, bacterial contamination may be unacceptably high. The least reduction is achieved in cases where sewage is discharged untreated or where treatment is restricted to the primary and secondary level (Ellis 1989).

Venice Lagoon is unusual insofar as there is potential to impact human health through combined exposure routes some of which are not immediately obvious. For example, transport in Venice is predominantly by boat. Disturbance of the water by hand propelled or motorised vessels may create aerosols. Aerosols of seawater have been demonstrated to contain enriched levels of bacteria, viruses and toxins (Blanchard 1989) and may constitute a health hazard for those living near the water as well as individuals in closer contact with it. Recent research (Plusquellec *et al.* 1991) has shown a 24-720 fold bacterial enrichment in the neuston as compared to subsurface waters. These workers concluded that this might increase the direct risks of water contact and also lead to high contamination levels in aerosols.

Remediation of the problem in Venice Lagoon will require careful planning to ensure that it is adequately and permanently addressed. The potential for mis-targeting efforts may be illustrated by the current strategy employed in the UK to meet EC standards. In the UK on a regional basis, compliance with the Bathing Water Directive has been found to vary between 0% and 95% at the 403 sites eventually designated

(Jones *et al.* 1990). The costs of bringing non-complying waters up to standard have been estimated at around £1000 million and much discussion has centred on the cost benefit analysis in the absence of robust epidemiological information. This equivocation is evident in the recently published UK proposals for Statutory Water Quality Objectives where it is suggested that the standards are applied on an interim basis (NRA 1991b).

The UK, nonetheless, seems committed to a dual policy. Installing primary and secondary treatment facilities in polluted estuaries such as the Tees and Mersey will improve aesthetic quality but will not enhance bacterial quality. In coastal regions, compliance with Directive EC/76/160 is hinged on a programme of long sea outfall construction such as that underway on the Fylde coast (Head *et al.* 1990). This will not leave much flexibility of response if the standards should be further tightened and will inevitably result in further capital expenditure (see: CES 1990). In neither case does the policy consider marine waters as anything other than an extension of the treatment process, where according to conventional wisdom, bacteria and viruses are rendered inactive by the combined effects of salt water and UV irradiation.

It is entirely possible that more stringent standards will be introduced in the future. Increasingly, the behaviour of bacteria and viruses in sea and estuarine waters is recognised as departing from the classical assumptions. Research has shown that sewage pathogens can survive for much greater periods in seawater and in sediments than had been supposed (Colwell *et al.* 1985; Colwell 1988). Some workers have suggested that the enterococci may be better indicators of sewage contamination than the coliforms (Pettibone *et al.* 1987). The Gram-negative bacteria in particular can survive in viable but non-culturable form which may lead to a serious underestimation of the degree of contamination using conventional techniques (Davies & Evison 1991). In an elegant experiment using labelling techniques, Garcia-Lara *et al.* (1991) found that loss of culturability of sewage bacteria was an order of magnitude greater than their rate of disappearance from culture. Under some circumstances in estuarine water, faecal indicator bacteria may actually increase in number (Rhodes & Kator 1988). Even the re-use of grey water has come under scrutiny recently due to the health risks associated with the pathogenic bacteria that it contains (Rose *et al.* 1991).

The utility of the conventional indicator organisms is also under scrutiny since they may not accurately reflect the true environmental stability of associated pathogens. Hurst (1989) notes that enteric viruses are known to persist in water for considerable periods of time (Johl *et al.* 1991; Olphen *et al.* 1991). As a further complication antibiotic resistance has been demonstrated in *Salmonella spp.* and coliform bacteria isolated from sewage (Stelzer & Ziegert 1988) and from sewage polluted waters (Al-Ghazali *et al.* 1988; Umaran *et al.* 1989; Morinigo *et al.* 1990). Egidius (1988) notes that acquired bacterial plasmid linked resistance may be transferred from sewage organisms to entirely different species. The recently identified importance of viruses in marine environments (Bergh *et al.* 1989) and the relative environmental stability of enteric bacteriophages (Cornax *et al.* 1991) also raises the possibility that resistance and other genetically encoded properties may be actively transferred to natural host populations of bacteria. The implications of this for marine systems remain unknown.

Similar considerations apply to the quality of shellfish waters under EC Directive 91/492/EEC due to come into effect on January 1st 1993. This will impose rigid conditions on the harvesting of shellfish such that on entry to the market, faecal coliform counts are below 300 per 100g of flesh and intravalvular fluid. Harvesting of fish with an initial coliform level above 6,000 faecal coliforms will be prohibited. In the majority of UK shellfisheries meeting this standard will entail conditioning by depuration or relaying. In ten areas, prohibitions will be required. Given the levels of coliform bacteria in the waters of Venice Lagoon, limits in animals harvested from the active shellfisheries seem likely to exceed the regulatory limit, and may be entering local markets in substantial quantities without adequate monitoring or conditioning.

Molluscan shellfish are well identified as vectors of bacterial and viral pathogens (Sato *et al.* 1992). Again the utility of the conventional pathogenic indicator organisms is in some doubt (Pancorbo & Barnhart 1992). In particular, the viruses may not be adequately monitored. Bacteriophages were found to accumulate more quickly than coliforms by *Mercenaria mercenaria* (Burkhardt *et al.* 1992). It has been established (Mesquita 1988; Power & Collins 1989) that bacteriophages are eliminated more slowly from *Mytilus edulis* than are enteric bacteria. Studies of New Zealand cockle beds led Nicholson *et al.* (1989) to conclude that the absence of coliform bacteria was not a reliable index of the absence of sewage derived poliovirus. In addition, there may be considerable problems in the assay of virus contamination of shellfish (Pietri *et al.* 1988).

Hence, even if the provisions of EC Directives 76/160/EEC and 91/492/EEC are fully adhered to, there is growing uncertainty as to whether they will provide adequate protection of the environment and human health. Growing doubts about the acceptability of discharging untreated or partially treated sewage to sea suggest that a more precautionary approach is required.

Full tertiary treatment would undoubtedly go a considerable way to removing microbial contamination. Many of the conventional sludge treatment processes inactivate viruses and bacteria (Hurst 1989; Lee *et al.* 1989). Stabilisation ponds can also reduce pathogens to extremely low levels (Ellis 1989). Quaternary treatment to produce a highly polished effluent is also possible. Filtration methods or irradiation with UV light are potentially suitable (Fahey 1990). There is a growing awareness, however, that chlorine is unsuitable as a final disinfecting agent due to the damaging effects of discharges upon receiving waters (Szal *et al.* 1991).

Chemical Contamination. The results shown in Figure 2 show the range of chemicals present in some sewage effluents sampled in the UK. Chemical contamination may derive from both domestic and industrial sources. Estimates have shown that appreciable quantities of metals enter sewage from domestic sources (Moriyama *et al.* 1989). The domestic contribution of 129 priority chemicals to sewage has been evaluated by the US Environmental Protection Agency (Hathaway 1980). This study notes that domestic product ingredients are often not well defined and that the relative contributions of domestic and industrial sources for some chemicals are difficult to evaluate.

Industrial discharges are undoubtedly a major source of organic and inorganic contaminants to sewage. In the UK, industry is actively encouraged to connect to the public sewer network as part of generalised pollution control initiatives (Belshaw & Fisher 1992). The adequate treatment of sewage to produce a microbiologically satisfactory effluent is contingent upon the existence of routes for the utilisation of the materials, particularly sludges, which intensive treatment produces. The presence of contaminants has been identified as a major restriction on the constructive use of sewage sludge (Glegg 1991). As noted by Rogers *et al.* (1989) organic chemicals in sewage may be resistant to breakdown and undergo partitioning onto sludge solids. In turn, this may lead to the elevation of organic chemicals in the sludge amended soils (see: Wild *et al.* 1990). Some groups of chemicals may be present at high concentrations. The linear alkylbenzene sulphonates and alkyphenol polyethoxylates may be present at part per thousand levels. Dichlorobenzenes and permethrins are likely to be found at concentrations up to 50ppm (Rogers *et al.* 1989). Lower, though significant (1-10 ppm dry weight) amounts of PCBs, chlorinated pesticides, and PAHs are also likely to be present (Rogers 1987).

While the partitioning of chemical contaminants to sludge has been widely studied, the chemical content of effluents has only recently begun to attract scrutiny. In a study of the chlorinated effluents in three publicly owned treatment works in New Jersey US, 323 chemical compounds were isolated and identified using on-column GC/MS. A further 91 could not be characterised. The analytical method used was also found to play a significant role in optimising the chemicals isolated with some chemicals being more amenable to LC/MS methods. The on column injection is thought to minimise sample degradation due to heated injection port linings resulting in the detection of larger numbers of compounds. In the New Jersey study a variety of compounds were isolated including organic acids, alcohols and surfactants, aldehydes and ketones, esters, halogenated compounds, heterocyclic compounds, hydrocarbons, phenolics, plasticisers, polynuclear aromatics and steroids (Clark *et al.* 1991).

The method reported here will necessarily resolve different chemical groups to those in the US study. Moreover, none of the effluents analysed were chlorinated prior to discharge which could be expected to result in qualitative differences. The traces shown in Figures 2a & b indicate that the effluents are variable and complex. In common with the US study, analysis reveals that many non-regulated pollutants are being discharged to surface and coastal waters, and those plants handling industrial wastes have increased numbers and quantities of synthetic organic chemicals in the discharge.

These findings throw into serious doubt assertions by water undertakings that it is possible to treat industrial effluents to an appropriately high standard. Particular problems may result in areas where a high level of water re-use is practiced. For example, in the UK, indirect re-use from rivers receiving sewage discharges affects some 30% of water supplies and some abstracted supplies may comprise 60% treated sewage effluent. (Lester 1990). Few studies have been done to evaluate the extent of this problem but Richardson & Bowron (1985) identified some pharmaceutical compounds which were not degraded by treatment and which could persist through treatment processes to the potable supply. The effects of these materials on receiving waters requires urgent evaluation.

CONCLUSION

The microbiological quality and chemical quality of sewage discharges are two key determinants of potential environmental impacts. Where sewage is discharged untreated into enclosed areas such as Venice Lagoon, microorganisms may be present at dangerous levels. Where discharge is direct to coastal waters as in the UK, breaching of microbiological standards may result in diminished amenity value and the shellfishing industry may be threatened. The utility of the standard indicator organisms for both bathing water quality and shellfisheries is questionable given the differential survival of various pathogens present in sewage. The interaction of sewage bacteria with natural populations may result in the transfer of genetic information. The significance of this is unknown. Realistic future environmental protection will involve the discharge of pathogen-free effluents. This will require tertiary treatment and in some cases quaternary treatment.

Sewage discharges are also an unregulated source of chemical contaminants into aquatic environments. This has serious implications for the treatment of industrial wastes at sewage treatment plants and in particular for the basing of pollution control strategies around encouraging industries to connect to the sewerage system. In many cases it is probable that chemical contaminants are not treated but are simply diluted in treatment works. The domestic use of chemicals is also a contributor to effluent contaminant budgets and the use of critical chemicals in the home needs to be examined on a systematic basis. Realistic environmental protection from the effects of sewage discharges will only be achieved by a fundamental reappraisal of the role of receiving waters as part of the treatment process based upon the adoption of a highly precautionary stance towards the potential environmental effects.

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