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URBAN DISCHARGES AND RECEIVING WATER QUALITY IMPACTS

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BIOLOGICAL ASSESSMENT OF EFFECTS OF COMBINED SEWER OVERFLOWS AND STORM WATER DISCHARGES

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ABSTRACT.

The biological effects of discharges from combined or separated sewer systems are difficult to assess or to predict due to variabilities in concentrations, environmental conditions, morphometry, susceptibility of organisms, seasonality and other factors. A general discussion of the problem results in an outline of two approaches. The first approach is to generalize field experiences and some examples of results are presented and explained. Notably site-specific conditions are shown to overshadow effects of discharges for many of the biological components in the ecosystem. The second approach is to perform toxicity tests which account for the temporal and spatial scales encountered in the field and for the relevant concentration levels.

KEYWORDS

Biological effects; bio-indicators; acute toxicity; chronic toxicity; combined sewer overflows; storm water; multivariate analyses; temporal scales; spatial scales.

INTRODUCTION

Increasingly waste-waters are being treated before discharge into receiving waters. Concomitantly the relative contribution of the remaining sources of pollution increases and more attention to their effects is paid in research and control. Among these sources the discharges from combined and separated sewer systems are frequently important, especially near urban or industrial areas. The effects of combined sewer overflows or storm water discharges upon the receiving waters are inherently transient in nature, and it has been recognised that physical and chemical characteristics of the discharges or the receiving water as such provide limited insight into the potentially detrimental effects on the level of populations of aquatic organisms or of the ecosystem. Mathematical modeling has contributed significantly to the insight into the frequency distribution of physico-chemical effects, but biological effects are still poorly amenable to prediction by modeling. Hence techniques, or rather approaches to assess harmful effects on the biota, are much in demand as the loss of function of the receiving water at least partially is related to the biological quality.

OBJECTIVES

The objective of this paper is to discuss and illustrate how potential harmful effects of CSO's and storm water discharges can be assessed at the population and ecosystem level and predicted by appropriate tests, the use of bio-indicators and the generalisation of field experiences.

DISCUSSION OF THE PROBLEM

Various and complicated problems are encountered when this task is considered. The complexity is due to:
- the variability in time, space and intensity of the loading and the concomitant effects
- the variety of processes affecting water quality determinants and the wide range of time constants for these processes
- the complex and variable composition of the discharged polluted water
- the differences in susceptibility to adverse environmental conditions for the various organisms living in water or sediments in relation to their specific life cycles, habitat selection, feeding habits etc.
- the seasonal and other variations in environmental conditions and the natural response of populations and ecosystems to such variation even in the absence of any pollution
- the variability due to differences in morphometry, sediment and water properties, flow regimes, dilution rates and other relevant site-specific conditions.

These factors exert their influence upon the ultimate systems response in an interactive way. Some of these interactions are shown in Figure 1 in which the spatial and temporal scales of effects in receiving waters are shown qualitatively. These effects are the result of the interaction of processes in the receiving water (transport, sedimentation, decay) and of response times of species or communities. The figure is far from complete but it illustrates that effects related to dynamic, high rate processes require a high frequency sampling or continuous monitoring in the vicinity of the outlet, whereas more chronic effects associated with slow processes can be observed more or less apart from the individual storm event and on a wider spatial scale. From this it will be self-evident that biological indicators for immediate or more chronic effects should be selected on the basis of their generation time as a first criterion.

Further it will be clear that differences in flow conditions will dominate over other factors. This is already true for non-polluted systems: running waters harbour quite different communities than stagnant or semi-stagnant systems. Also the recovery of a system after a disturbance will depend strongly on the flow conditions and the opportunities for recolonisation from upstream areas.

Biological effects that can be expected to occur include:
- wash out of organisms with the water and scour of organisms out of the sediments (comparable to catastrophic drift);
- toxicity effects of micropollutants;
- reduction of DO resulting in die off of organisms;
- turbidity, reducing primary productivity, but also
- stimulation of plant growth by nutrient input, etc.

The duration and level of the adverse concentrations may lead to acute toxicity effects, but generally chronic effects due to prolonged exposure at low concentrations of micropollutants are more likely to occur than acute consequences. Low oxygen concentrations, however, can be expected shortly after an overflow event and can be very harmful for fish, particularly when these conditions are prolonged. Even prolonged exposure of fish to moderate deficits has been shown to have comparable effects. This has resulted in recommended standards for DO as a function of exposure time in Denmark (Hvitved-Jacobsen, 1986), which is probably too sophisticated for practical management. More important is the concept to relate standards to recurrence time as proposed by the same author. This concept is important in relation with the generation time of organisms: a long generation time requires a long return period for adverse conditions to become acceptable.

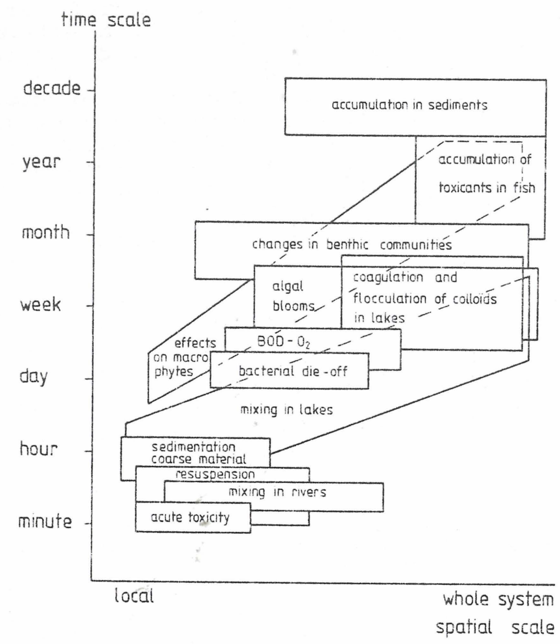

Figure 1. Relation between the rates of processes and the spatial scale of the effects (from: Aalderink and Lijklema, 1985).

Ideally, impact studies would require controls in both time and space (Green, 1978). This is so because evidence for an effect must be based on changes that occurred in the test area, but not in the control area. Such a set up would allow for factorial design. However in the real world generally impacts have already occurred near the outlet, unless a new discharge is being studied. Hence a rigorous statistical approach is not possible. Gradients observed in the biotic community in the affected area may be used to interpret impacts.

As to the choice of indicator or test organisms, it will be evident from inspection of Figure 1 that more permanent effects can be expected to show up in the bottom dwelling organisms because in the sediments pollutants tend to accumulate and adverse conditions are more permanent. Short term effects would show up predominantly in short living organisms in the water phase. Fish, with their complex interactions in the food web and mobile character, would exhibit both acute and chronic effects. It should be noted that effects on the level of individual animals such as small changes in metabolic rates or yields may remain imperceptible, but they may show up in the next level of integration: population dynamics (Halbach, 1984). Extrapolation of this observation leads to the conclusion that no-effect levels will be even lower for the ecosystem. Hence interpretation of field data or test results on the level of isolated organisms or species should be made with caution.

TWO APPROACHES

Considering the variability in time, space, effluent composition, the site-specific conditions and the lack of detailed knowledge of the interacting physical, chemical and biological processes, no uniform, simple and unbiassed methodology is available to assess, to evaluate and to predict harmful effects on the biological system. In the opinion of the authors the problem can be approached from two different sides, each with their limitations but mutually more or less supplementary.

The first approach: Evaluation and generalization of field experiences.
The basic idea is that from a large data set of effects upon several organisms, including spatial and temporal gradients and different combinations of sewer system, morphometry of receiving water, season etc. a general insight can be obtained into the extent and duration of impacts. The conditions to be expected at a specific site can be inferred from the existing experience by analogy and compared to what is desirable and acceptable. Alternatively the position of an object in the reference framework can be assessed and compared and the changes to be expected from management measures indicated. The task involves the evaluation and interpretation of large data sets and the use of techniques that allow the discrimination between cause-effect relationships. Such an analysis also may indicate which organisms are suitable bio-indicators for impact analysis.
The second approach: Direct tests with indicator organisms.
This approach is the classical testing of the effect of an effluent and/or the suspended solids on a test population or individual organism. This includes bio-assays in which the relative potency of a solution is compared with the effect of a standard solution (such as the assays used in eutrophication studies) and, more important here, toxicity tests in which the degree of response produced by exposure to an effluent is being measured.
Both approaches will be discussed in some detail, for the first also some original data will be presented.

Evaluation and generalization of field experiences.
External influences on an ecosystem might be detected by their effects upon the functioning and/or structure of the biological components. However, generalization of observed effects is hampered by the naturally occurring differences in functioning and structure among (communities of) organisms due to distribution and ecosystem development patterns. These patterns are primarily determined by master factors as, for instance, seasonality, current velocity, salinity, morphometry, acidity and sediment composition.

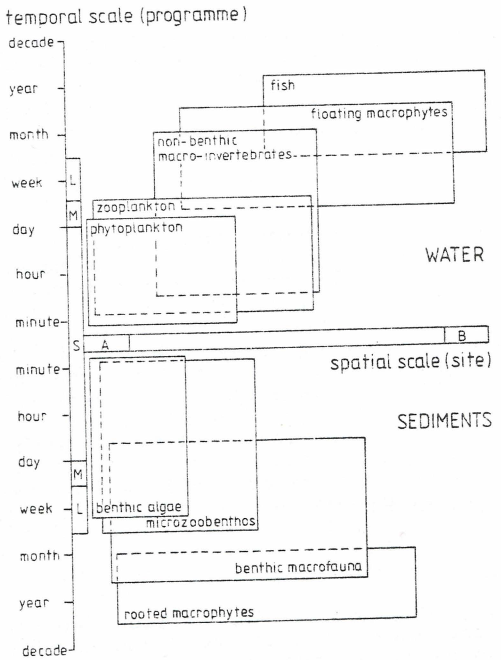

Figure 2. Relation between temporal and spatial scales of the effects on the various biological components, compared to the sampling programme and site.

An extensive study from 1985 to 1987 included a wide variety of localities throughout The Netherlands (> 60), selected on the basis of type of sewer system and receiving water. At each locality three sampling sites have been selected:
site A: in the immediate vicinity of the overflow;
site B: at some distance of the overflow, but within its sphere of influence;
site C: a reference site in the same type of water, close to, but not influenced by the overflow.
As effects of overflows on biological objects are expected to be reflected in different time scales (Figure 2), depending on the type of organism studied and their habitat (free-floating or planktonic versus rooted or benthic), the sampling programme included:
programme S(hort): immediately after an overflow event;
programme M(edium): one to four days later;
programme L(ong): one or two weeks after an overflow event;
programme B(background): at least one month after an overflow event.
Depending on morphometry and other characteristics of the sampling sites (habitat selection) and on the life-cycle of the organisms, the following biological objects have been studied: phytoplankton (S & M), microfauna (zooplankton: S, M & L and zoobenthos: L), epiphytic diatoms (B), macro-invertebrates (L & B) and macroflora (B).

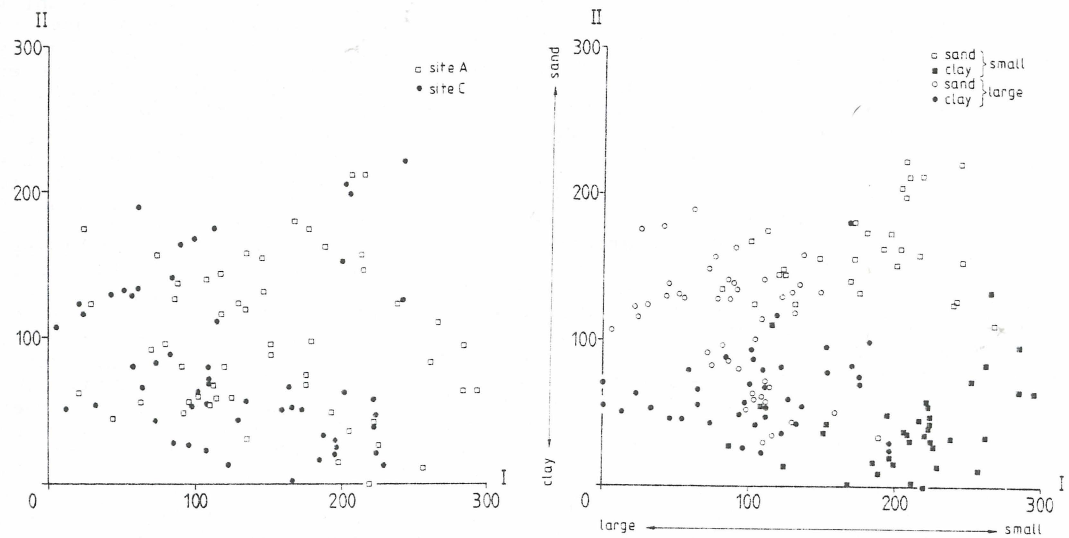

Figure 3a (left). Ordination diagram of sampling sites A and C on the basis of macro-invertebrate species composition and abundance.
Figure 3b (right). Ordination diagram of sampling sites A, B and C on the basis of macro-invertebrate species composition and abundance.

Clustering (TWINSPAN) and ordination (DECORANA) of the sampling sites on the basis of similarity in species composition and abundance proved to be useful in visualizing the effects of overflows on the biological objects mentioned above. Only the results of the macro-invertebrates investigations are used here to illustrate this. Figure 3a shows a diagram of the A and C sampling sites in (semi-)stagnant waters, ordinated on the basis of their macrofauna species composition and abundance. In contrast to expectations, the A and C sites are distributed randomly throughout the plot, indicating that the ordination of the sites (including the B sites) is not determined primarily by the presence of an overflow. However, in Figure 3b it can be seen that ordination of all the sites is largely determined by the size of the receiving water body (first principal axis) and the sediment composition (second principal axis). So the influence of overflows on macro-invertebrates is only of secondary importance. As site-specific conditions (morphometry and sediment composition) are so important, harmful effects can only be detected when receiving waters are already subdivided in different types.

In general, the smaller or more isolated the water body, the more pronounced differences in species composition between the sites are. In large closed waters organic matter, heavy metals etc. accumulate in the deeper parts, and there the first effects will be found.
As far as toxic components are not released or of secondary importance, traditional biological assessment methods can also be used (Pantle and Buck, 1955; Woodiwiss, 1964; Moller Pillot, 1971; Sládececk, 1973; Gardeniers and Tolkamp, 1976). These methods are based upon (groups of) species indicating the saprobic level, traditionally associated with measurements of the D.O., biological oxygen demand and ammonium.
After an overflow event restoration of water quality is (partly) possible, particularly in the water phase. Recovery will largely depend on inoculation of the water body with species. If the water body is isolated and difficult to access inoculation is prevented. Soil conditions will deteriorate further after each new event. Due to mineralization of organic matter oxygen consumption will increase and heavy metals accumulate.

As indicated previously, the different biological components in an ecosystem will react differently upon external changes. To illustrate the differences in reaction (time) upon overflow events one type of surface water has been selected to study the effects of overflows upon the different biological components of the system. A pond frequently disturbed by overflows (Loenen) has been studied intensively during one year (1984) using a fortnightly sampling programme. A non-disturbed pond in the neighbourhood has been used as reference (Apeldoorn). Results of these investigations have been published by Willemsen and Cuppen (1986) and by Roijackers and Ebbeng (1986). Effects on macro-invertebrates were not pronounced. Important effects were physically induced, e.g. passive transportation by the incoming water masses (invertebrates, plankton, benthic organisms) or changes in turbidity (algae). The recurrent overflows in the detention pond prevented maturation of the ecosystem and community structure was minimal. The communities of epiphytic diatoms in particular showed an architecture accomodated to the resistance of flushing, whereas the epiphytic diatom communities in the reference pond showed a much more pronounced architecture. Discrimination on saprobic degree was possible, but fluctuations through time were barely detected (Figure 4: top). Phytoplankton reacted mainly upon changes in light climate as nutrients were never limiting. After an overflow event phytoplankton and turbidity causing colloidal matter was flushed and the water mass became very transparent for a few days. Due to the higher irradiance level phytoplankton biomass increased and phytoplankton species were characterized by their high turnover rate. Phytoplankton was not suitable to indicate saprobic degrees (Figure 4: bottom). Zooplankton clearly reacted on water movements and input of organic matter (Figure 4: middle). In the case of zooplankton and macro-invertebrates the importance of inoculation from the shore vegetation (a temporary refuge) was clearly indicated. From this study it was clear that macro-invertebrates, macrophytes and epiphytic diatoms reflect the long term effects of overflow events, whereas algae and micro-invertebrates reflect short term effects (Figure 2).

On the basis of the above-mentioned studies a suitable biological component can be selected for field observations, depending on the type of water and objective of study. Also these studies show which species are suitable indicator organisms that can be used in laboratory studies among which bio-assays are most popular. In future, however, more attention should be paid to larger scale laboratory experiments, in which natural situations are reflected better and more complex communities can be studied (Kersting, 1984a, b). Enclosure experiments proved in several other situations to be an excellent tool in testing the effects of pollutants under natural conditions.

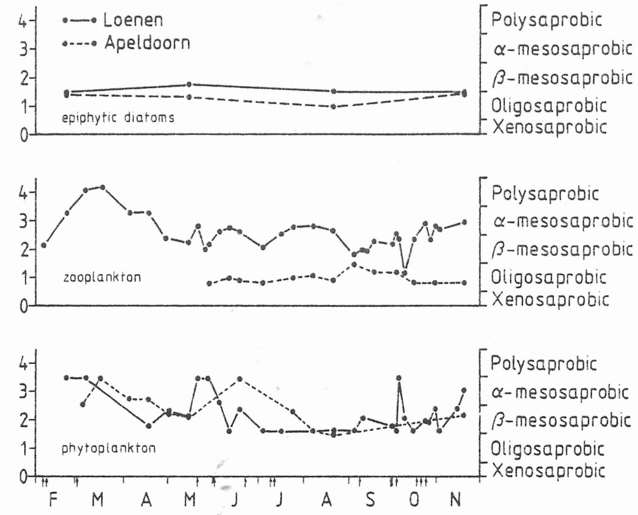

Figure 4. The saprobic level according to Sládecek (1973) for a pond in Loenen frequently disturbed by overflows and a reference pond in Apeldoorn (from: Roijackers and Ebbeng, 1986).

TOXICITY TESTS, BIO-ASSAYS

In order to be relevant for field conditions toxicity tests and bio-assays should represent as well as possible:
a) the (variations in) exposure time for the different organisms
b) the (variations in) concentrations of pollutants and concomitant physico-chemical conditions
Most standard tests do not take into account time-concentration relationships: typically an organism is exposed to a fixed concentration for a defined time period. In the environment the concentration will decrease due to degradation, dispersion, sedimentation etc. The rate of decrease will be different for the various constituents in the waste-waters. This is difficult to simulate in a laboratory test.

Exposure time.
Besides the removal rates of pollutants and the fading away of adverse conditions of pH, low DO etc., the exposure time for a biological test should take into account the flow conditions in the receiving water and the mobility of the organism used as a bio-indicator. In running waters the exposure time will be short for mobile organisms (fish), somewhat longer for passive organisms moving along with the water (phytoplankton, zooplankton) and short for aquatic weeds. In stagnant linear waters (ditches, canals) the exposure time will be longer, especially for sessile organisms, while exposure time is long for all organisms in closed water bodies (ponds, lakes). In the sediments the exposure time will be long, except for readily degradable matter.
From the above it follows that tests on organisms living in the water phase should primarily be acute toxicity tests. Such tests are frequently mortality tests, in which the median lethal concentration LC50 is estimated after a preset, fixed time of exposure. This concentration causing 50 % mortality has the least variability in the concentration-response curve. The exposure time is generally 24, 48 or 96 h. Other effects are sometimes measured instead of mortality, such as suppression of growth in algae, immobility or loss of equilibrium in invertebrates. The expression EC50 is used in such cases: median effective concentration.

Although toxicologists generally prefer flow-through test conditions, a static test may be more representative for overflows from sewer systems. In such tests the solution (effluent) is added to the dilution water in the desired proportion after which the test organisms are added without any change of water during the test. This exposure system allows degradation, build up of metabolic products, adsorption, volatilization and depletion of oxygen to occur. All this is more realistic, but also more difficult to document and standardize. Of particular interest is the probability of synergistic effects with low oxygen concentrations. For instance the LC50 for 96 h exposure of rainbow trout has been shown to decrease with decreasing DO concentration (Thurston, 1981, in Russo, 1985).

For CSO's the recurrence time is generally long. This means that the fate of organisms with a short life cycle is not very interesting and mainly higher organisms should be tested. For isolated systems where recolonization is a problem this is not necessarily true. For storm water discharges the recurrence time is much shorter, hence short living organisms may become important; certainly as they serve also as a food source (algae). In the case of algae stimulation of growth may occur as well as a consequence of nutrient addition. A bio-assay may then indicate the potential for growth stimulation, but a chemical analysis of nitrogen and phosphate may be adequate.
As discharges generally occur during all seasons, it seems appropriate to pay special attention to the early life stages, eggs etc., because these tend to be the most sensitive, probably due to the absence of a detoxification mechanism.
Organisms living in sediments are exposed to more prolonged harmful conditions. Except for readily biodegradable and volatile compounds, the pollutants associated with settling particles will exert their influence on benthic organisms for periods which well may exceed the recurrence time of overflow events. Hence chronic tests seem appropriate. A major problem with chronic toxicity tests in general is that they are expensive. In the case of CSO or storm water it is also difficult to have the disposal of a supply of test solution of constant and well defined composition. These are reasons to consider methods of extrapolating acute toxicity tests. For this purpose the concept of an application factor (AF) has been developed, which is the Maximum Acceptable Toxic Concentration (MATC) divided by LC50. Instead of MATC also the range in which it lies: No Observed Effect Concentration (NOEC) to Lowest Observed Effect Concentration (LOEC) is used to define the AF value. The assumption is that the ratio between the chronic test values (MATC, NOEC and LOEC) and the acute toxicity value LC50 is relatively independent of the species tested for a specific chemical (Rand and Petrocelli, 1985).As a consequence the tests need not to be performed with long living organisms for which the test period may be as long as one year. Instead the tests can be made with for instance Daphnia-species to assess the AF value, after which only an acute test with other relevant species is needed. For application to benthic organisms exposed to quite different substrates than **Daphnia** this comparison fails of course. Furthermore no information regarding the independency of AF for complex effluents is known to the authors.
Extrapolation of Early Life Stage tests to MATC has proven to be succesful in many cases (McKim, 1985) for individual toxicants and some support is available that also for mixtures, including sewage, MATC can be derived from ELS tests. Hence this direction seems to be promising for further development. A major problem may be that the concentration in effluents is insufficient to assess the LC50 value properly, whereas dilution in the receiving water is not such that chronic effects can be excluded off-hand. Concentration of effluents will affect their physico-chemical properties.

Concentration effects.

Test conditions should represent as well as possible:
- the variations in composition of the effluent;
- the variations in volume of the discharge and hence the initial dilution in the receiving water;
- the temporal and spatial gradients in concentrations in the receiving water.

The maximum concentration in the receiving water will be that of the non-diluted effluent as it may push aside stagnant water or is mixed with insignificant flows in a receiving stream. With increasing time and/or distance the concentrations will decrease. There is a distribution in effects to be expected related to the distribution in concentrations and exposure times. As stated in the previous section, the decreasing concentrations in time due to reactions may be simulated by performing stationary tests. Variations due to spatial gradients and mixing can be simulated by using dilution series. In each of the individual test solutions also the temporal effect of degradation etc. will occur to a certain extent.
Generally the maximum toxicity will be the most important quantity to assess. For this it may be necessary to know the seasonal variation in flow of the receiving water and the most sensitive life stage of the species to be tested. Usually the most critical conditions for these two variables will not coincide; for instance the most sensitive life stage in the spring and low flow conditions in the (late) summer. Low flow and high temperature will frequently go together and may induce low DO levels. This all accentuates the importance of arranging the test as much as possible in accordance with the local, site-specific conditions.
The treatment of the variation in effluent composition and strength necessarily must be simple. The more toxic conditions deserve most attention, but it is not practical to store effluents and perform tests with the samples with the highest concentrations. It seems to be more appropriate to test one or two sampled effluents in a dilution series and to extrapolate the measured effects to the higher concentration range on the basis of a distribution of concentrations. This distribution may follow from chemical analyses of a greater number of samples. Extrapolation to the geometric mean of the highest quartile of concentrations might be a reasonable approach. Alternatively model results can be used but for quality the results are still rather poor. Of course a great number of constituents in the effluent plays a role and the variation in concentration will be different for each variable. So the extrapolation suggested should be applied to the variable(s) which is (are) suspected to dominate the effects. This can be based on the results of chemical analyses and data on the toxicity of individual chemicals.
Results of acute toxicity tests performed according to the general guidelines discussed above, should then also serve to assess chronic toxicity. For most chemicals studied the chronic toxic concentration is higher than 1 % of the acute toxicity concentration. Macek (1985) reports that the limited data on complex waste waters and treated industrial effluents provide evidence that an application factor in the range of 0.05-0.1 would be a reasonably good estimate.

Field testing.

Although on-site testing in principle would be more informative and realistic in several respects, it is practically difficult due to the stochastic nature of overflows. For a limited number of organisms however, comparative tests in the field can be set up. For instance in streams, flow-through enclosures with the test organism can be placed upstream and downstream of a discharge. For a more or less frequent discharge from a separated sewer pipe such an arrangement can be very useful. Sessile plants and animals and (caged) fish can be used.

MATHEMATICAL MODELS

These will be discussed in other contributions in this volume. Here it suffices to state that, although ecological effects cannot yet be predicted properly by models, a valuable contribution to the insight in the fate and effects of specific toxicants can be obtained from mathematical models. They complement other information and help to organize studies and the analysis of their results. In this respect the models TOXIWASP and EXAMS should be mentioned, see e.g. Mulkey et al. (1986).

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