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The Value of Aquatic Model Ecosystem Studies in Ecotoxicology

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CONTENTS

SUMMARY

The process of risk assessment of substances aims at safeguarding the integrity of complex environments and ecosystems. In this context, a No-Effect Concentration for environmental organisms needs to be predicted (= PNEC) on the basis of a limited amount of ecological and ecotoxicological data available. Most of the substance-specific data have been generated on single species under laboratory conditions, and empirically-derived assessment factors are currently used for the extrapolation to the real environment. It is the purpose of this report to explore in detail the value of aquatic model ecosystem studies in predicting the effect of substances in the "real world" ecosystem.

The relevant scientific literature was thoroughly screened and the various types of studies found were described separately for the three broad groups of studies, i.e., static freshwater, flowing freshwater and marine systems. Large differences exist among the reported studies concerning the test conditions chosen, particularly location, duration, size and complexity. The Task Force concluded that it seems inappropriate at this stage, to recommend a single standard test design or a set of designs. Each study should be tailored to address the specific issues or data requirements that have arisen from earlier stages of testing.

To enable safe concentrations to be forecast by means of assessment factors backed up by sound scientific data, a two-step procedure was followed:

- prediction from chronic single-species No-Observed-Effect-Concentrations (NOECs) to model ecosystem NOECs;

- prediction from model ecosystem NOECs to field NOECs (=PNECs).

To establish the potential usefulness and the role of model ecosystems in risk assessment, NOECs obtained from well-designed model ecosystem studies were compared with NOECs obtained from laboratory single-species tests on the one hand and with field studies on the other.

A database has been assembled containing high quality published information on the toxicity of substances in ecosystem studies and those from chronic single-species tests. Those ecosystem studies which provided values for both NOECs as well as the corresponding Lowest-Observed-Effect-Concentrations (LOECs) were selected following a critical review of the literature. From a total of 1108 data points only 248 studies fulfilled this criterion. They covered 34 substances. The data from single-species tests were extracted from the ECETOC Aquatic Toxicity (EAT) database, complemented with company and additional literature data of comparable quality.

The ratios between the most sensitive single-species NOECs and the most sensitive multi-species NOECs were compared, irrespective of ecological relevance. This was considered to be a reasonably conservative approach for the derivation of assessment factors. In the evaluation of model ecosystem studies for a particular substance, however, it is necessary to select, from the various endpoints recorded, the lowest one which is ecologically significant. Such evaluations, performed with three substances in this report, also demonstrate the high degree of conservatism of the above assessment factor.

For the prediction from chronic single-species NOECs to model ecosystem NOECs, the median value for the ratios (which ranged from 0.02-77.5 with log-normal distribution) was found to be 1.45 with a 90%ile value of 8.14. This suggests that an assessment factor of about 8 for the extrapolation from the lowest chronic single-species NOEC-value to a NOEC-value in a model ecosystem would be safe.

For the second step a comparison was made between results from model ecosystems and results from field studies. The conclusion was that results from the model ecosystem studies of sufficient complexity could be considered as realistic for the real world situation.

This means that an assessment factor of 8 is equally suitable for the prediction of a safe environmental concentration (PNEC) on the basis of chronic single-species NOECs.

1. INTRODUCTION

The environmental risk assessment of a substance is generally based on a comparison of its Predicted Environmental Concentration (PEC) with the Predicted No-Effect Concentration (PNEC). PNEC values are typically calculated from single-species acute or chronic laboratory toxicity tests using an appropriate assessment factor (US-EPA, 1984; OECD 1992; EEC 1996). It is assumed that where the PEC exceeds the PNEC (i.e., PEC/PNEC >1), there could be a potential for environmental effects. The process allows for, where necessary, a stepwise refinement of both the PEC and/or the PNEC independently from each other (ECETOC, 1993a; EEC, 1996). When the PEC/PNEC ratio exceeds unity and there appears to be a necessity of refining the PNEC, ecosystem studies are considered to be a suitable instrument for the derivation of a more realistic PNEC.

The principal purpose of model ecosystem studies in ecotoxicology is to provide data on the effects, and sometimes also fate, of substances under conditions which are more representative of the 'real world' than single-species laboratory tests. This is expressed in terms of greater realism concerning for example, exposure conditions and in the variety of data that can be collected, such as effects on several taxa examined in the same test. Furthermore, model ecosystem studies allow the examination of effects on endpoints based on functional or structural aspects at the ecosystem level. Thus these studies provide the opportunity to gain further insight into the ecological significance of the effects seen.

To explore in detail the value of the various model ecosystem studies in predicting the effect of substances in the environment, ECETOC established a Task Force with the following Terms of Reference:

- collate and critically evaluate the existing literature on biocoenosis studies;
- describe the techniques involved in biocoenosis studies;
- compare the test results obtained from experiments using biocoenosis and single-species approaches;
- evaluate the value and the consequences of using biocoenosis approaches to ecotoxicology testing.

This report is one of a series of ECETOC Technical Reports published in recent years that deals with the general and specific aspects of environmental risk assessment (ECETOC 1993a; 1993b; 1994a;

1994b; 1996). It considers the value of aquatic model ecosystem studies in the environmental risk assessment process, describing the various types of studies and their uses (Section 3), reviewing the possible extrapolation of results from chronic single-species studies to model ecosystems and from model ecosystems to the 'real world' (Section 4) and finally presents conclusions and recommendations (Section 5).

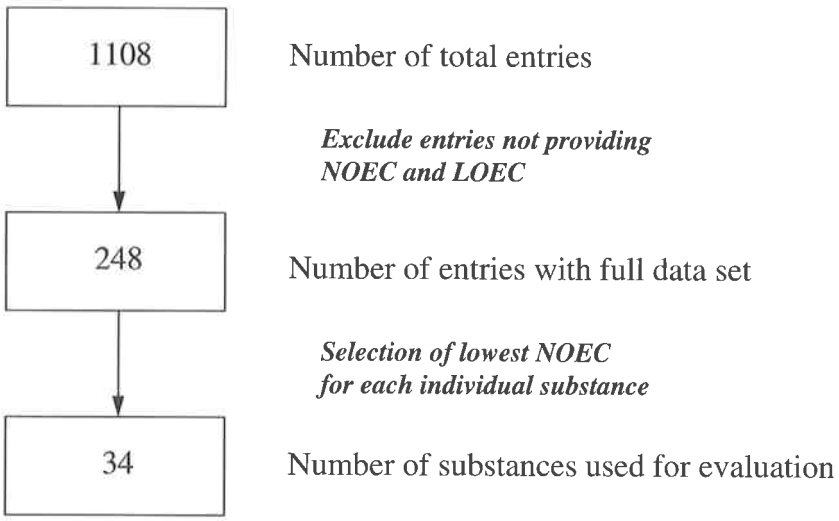
2. BACKGROUND

Various workshops were held and guidance documents were issued within recent years discussing test design and interpretation of results for freshwater model ecosystem studies (SETAC Europe, 1991; SETAC-RESOLVE, 1992; Graney *et al*, 1994; Hill *et al*, 1994). The results of these workshops were considered by the Task Force along with other relevant scientific literature.

The literature on model ecosystem studies was collected by electronic and manual searches. In view of the limited number of suitable terrestrial studies available in the open literature the report has been confined to a review of aquatic studies. The papers on aquatic studies were subdivided into three broad groups, i.e., flowing freshwater, static freshwater and marine systems.

Initially some 1108 literature references (Step 1 in Figure 1) were reviewed. These publications provided the basis for the analysis of the different test designs which have been developed for model ecosystem studies. The results of this analysis are summarised in Section 3 and reported in detail in Appendix A.

Figure 1: Process of Data Selection



For the quantitative analysis of the results of model ecosystem studies with those of single-species tests, however, the studies were in general only considered further if:

- they were well documented, published in peer-reviewed journals or in comprehensive, widely respected reviews;
- the data were supported by adequate chemical analysis;

- they reported both Lowest Observed Effect Concentrations (LOEC) *and* No Observed Effect Concentrations (NOEC);

- they were judged to be scientifically sound in design and execution (expert judgement).

Only 248 of the 1108 single model ecosystem studies fulfilled these criteria and were included in the database. For nine out of the 248 entries no reliable single-species values could be found. The remaining 239 entries comprised 34 different chemicals.

For the 34 chemicals identified above, chronic single-species toxicity NOEC values were extracted either from the ECETOC Aquatic Toxicity (EAT) data base (ECETOC 1993b) which has well-defined quality criteria for data acceptance or from other sources which were individually assessed for quality using criteria broadly in line with those of the EAT data base.

In some cases, where no chronic NOEC values were available, acute LC_{50}/EC_{50} values were taken and a factor of 10 was used to extrapolate from acute to chronic data (see Appendix D for detail). The approach was considered justified because of the generic aspect of this study and the additional conservatism in respect to the calculation of the NOEC ratios.

The relationship between the relative sensitivity of endpoints from single-species chronic toxicity tests, model ecosystem studies and field monitoring has been analysed in order to provide information on the extent to which the results of model ecosystem studies can be used to refine a PNEC derived by applying an assessment factor to the results of single-species tests. Ideally, such a comparison of the data of single-species tests to those of multi-species tests should be performed on the basis of threshold concentrations. Because test design normally does not allow the determination of a precise threshold concentration, comparisons are made on the basis of the NOECs.

Since the cost and effort involved in mounting model ecosystem studies usually limits the number of concentrations employed, dilution factors applied to most ecosystem studies range from 3 to 10 rather than from $\sqrt{2}$ to 2 as usually applied in single-species tests. Consequently the difference between the NOEC and the actual (unknown) threshold is generally greater in ecosystem studies than is the case for single-species tests. This potentially greater internal safety margin may provide for an additional factor of up to 8 in comparison to that of single-species tests. Hence the approach chosen resulted in additional conservatism.

All results refer to the substance itself; in the case of heavy metals the results are calculated on the basis of the cationic species.

A number of terms is used consistently throughout the report: The term *biocoenosis* is defined as an assemblage of organisms (plants, animals and bacteria) inhabiting a single biotope which interact with each other and their abiotic environment. It is synonymous with *community*. An *ecosystem* is defined as a natural unit consisting of a biocoenosis and its abiotic environment interacting to produce a stable system. *Model ecosystem* is part of the (natural) ecosystem comprising the main structural and functional parts of a real-world ecosystem but in a man-made structure. It is the last term which describes best the kind of studies which are reviewed in this report.

3. AQUATIC MODEL ECOSYSTEMS AND THEIR USE

3.1 GENERAL ASPECTS

The principal purpose of model ecosystem studies in ecotoxicology is to provide data on the fate and/or effects of substances under conditions which are more representative of the 'real world' than single-species laboratory tests. This is expressed in terms of greater realism concerning e.g., exposure conditions and in the variety of data that can be collected, e.g., effects on several taxa can be examined in the same test.

The choice of the test system (i.e., type of ecosystem to be used for testing) and the test design (i.e., location, size, duration and biological complexity) must be tailored for each study based on the existing knowledge of the fate and effects of the substance. It is therefore not possible, nor desirable, to define in advance details of the test systems or test design to be used. It is however, possible to indicate a number of more general aspects of test system and test design which should be considered when determining the type of study to be undertaken.

Three broad groups of model ecosystem studies can be identified: static freshwater, flowing freshwater and marine (usually static or with relatively-long replacement times). The variety of model ecosystem studies that have been used to assess the effects of substances within each of these three broad groups is large. Sections 3.2, 3.3 and 3.4 indicate the range and variety of these different model ecosystem studies and a more comprehensive description of them is given in Appendix A.

The principal aim of this report is to assess the value of model ecosystem studies in predicting the *effects* of substances in the environment. However, these studies can also provide useful information on aspects of exposure. Model ecosystem studies will, by their very nature, ensure that exposure is more realistic than in experiments carried out in less-complex systems.

There are various levels of biological organisation at which endpoints can be determined in an ecosystem. They range from effects on cells or organs of a test organism (i.e., sub-individual effects) via effects on individuals and on populations up to effects on community function and structure. The nature of the effects data required can be used to guide the choice of the test system and experimental design. The expectation is that as the size, duration and biological complexity of the test system increases so will the likelihood of detecting effects at the higher levels of biological organisation. Of course, there will be limits beyond which increasing size, duration and complexity do not bring concomitant rewards. In practice very large systems may reduce the chance to detect effects due to the difficulty of controlling variability between replicates as complexity of the test system

and duration of the study increase. As size and duration of the studies increase costs are also likely to rise. This should not be confused, however, with the enhanced predictive and explanatory power of large test systems. Due to their size these systems can accommodate fish and become excellent surrogates for natural systems.

Selection of the appropriate test system and experimental design must be based on a thorough knowledge of the capabilities of the various systems and the data requirements. Maximum value is likely to come from studies where relatively stable ecological communities are established in a replicated form and effects are examined based on population, community functional and structural endpoints. Figures 2, 3 and 4 illustrate the relationships between size, location, duration, type of community and biological complexity for static freshwater, flowing freshwater and marine model ecosystem studies reviewed.

One important element of all model ecosystem studies is the mode of application of the test substance. In general, it is desirable that this should reflect what occurs in the real world in terms of the rate, frequency and nature of application.

General chemicals and metals typically enter natural waters as components of effluent discharges (treated or untreated). Three exposure scenarios can be distinguished:

- short-duration spikes of contamination, for instance caused by accidental releases;
- intermittent contamination for example, by effluents from industrial plants with batch processes;
- continuous discharges, which is the case for most industrial and domestic sewage effluents.

Inland effluents are generally discharged into flowing fresh waters; model stream ecosystems have therefore often been used to examine their potential effects.

Discharges to the marine environment have been studied by both single additions to static systems and by continuous additions to model ecosystems with rather long residence times.

Pesticides under normal conditions of use may enter the aquatic environment by spray-drift or run-off following commercial applications to the land, though in some instances they will be deliberately applied to water (e.g., aquatic herbicides). Commercial applications of pesticides are typically of short duration, a maximum of hours, as single or intermittent events, and seasonal. Static freshwater model ecosystems have been most widely used to study the effects of pesticides.

Where spray-drift is the route of entry to be simulated it is desirable that the frequency and rate of application to the model ecosystem should be representative of commercial use. Typically model ecosystem studies of spray-drift have involved single or repeat "oversprays" of the model ecosystem at application rates extending from the commercial rate to rates that might represent the spray-drift onto a waterbody from an application to a crop some distance away.

Run-off following application of pesticides is a complex phenomenon. The duration and the nature of episodes vary depending upon rates of application, frequency of application, soil conditions, weather etc. In view of this complexity it is not possible to indicate in advance what might be the appropriate method of application of a pesticide in a particular study. Applications may range from single or repeat treatments with a slurry of soil-adsorbed pesticide to a static water model ecosystem (e.g., a pond), representing the result of a run-off event induced by heavy rain after a crop treatment, to a more or less continuous application of low concentrations of dissolved pesticide to a flowing water model ecosystem (e.g., streams) representing an input from tile drains.

When effects on non-target organisms of deliberate applications to aquatic systems are to be assessed it is important that the conditions of the study follow those recommended for commercial applications.

Dosing of the test substance into the model ecosystem should, as far as possible, simulate the 'real world' discharges/releases in terms of concentration, duration and other factors that may be relevant (e.g., presence of suspended solids and other dissolved organic matter).

An important element in correctly applying a substance in any model ecosystem study is to ensure, as far as is practicable, that once in the test system the substance has the same bioavailability as it would have in the real world.

Bioavailability can be influenced by many factors for example, water quality (e.g., pH, concentrations of suspended solids and dissolved organic matter) and the possible routes of uptake (e.g., via the water only, or via food and water). Many substances enter fresh and marine waters via waste water treatment plant effluents. In these situations tests should ideally be carried out under conditions that simulate the presence of the test substance in a treated effluent. However, in practice this can be difficult or impossible.

Model ecosystem studies will by their nature ensure that exposure is more realistic than will have been the case in experiments carried out in less-complex systems, but positive efforts should be made to maximise the realism of exposure. This enhanced realism, for example, in a plankton study should mean that exposure is carried out in the presence of realistic concentrations of dissolved organic

carbon and suspended solids and not in 'clean' water. In more complex tests, where for example, a sediment phase is present, it should mean that there is the opportunity for uptake from the sediment to take place if this is relevant.

It is apparent that more realistic exposures will result in more realistic effects than those seen in, or predicted from, laboratory single-species tests. It is also clear that in some instances realistic exposures in model ecosystems will result in increased toxic effects (e.g., where additional routes of exposure are present (Hermanutz, 1992)) and in others reduced effects (e.g., where availability is reduced by complexation with organic matter (McCarthy and Jiminez, 1985)). These more realistic model ecosystem studies also provide a better basis for judging the ecological significance of effects than do laboratory single-species tests.

3.2 STATIC FRESHWATER MODEL ECOSYSTEM STUDIES

Static freshwater model ecosystem studies have ranged from simple experiments in small indoor tanks containing a small number of pelagic species to studies in large outdoor ponds that closely resemble many aspects of a natural lake or pond.

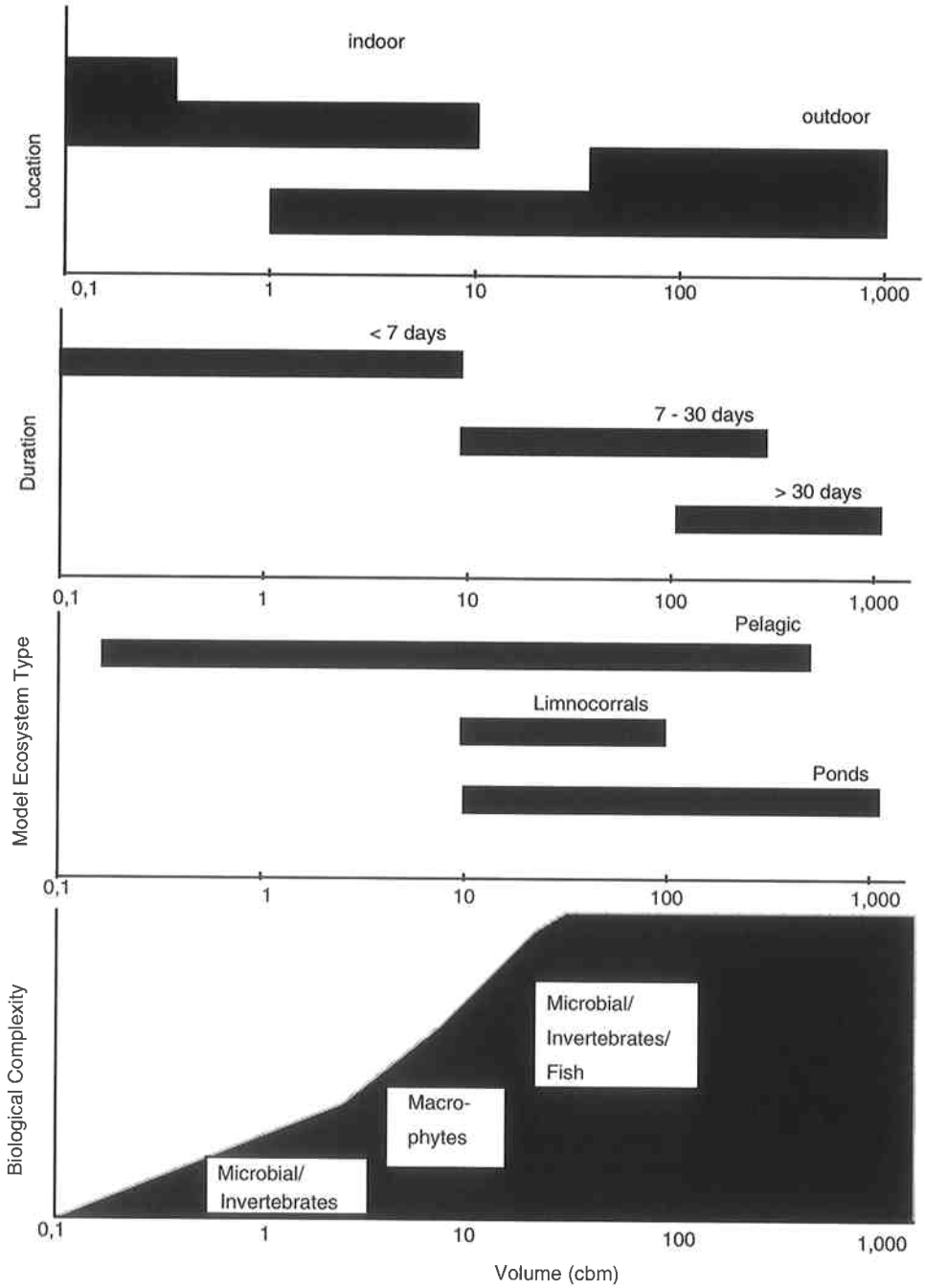
The smallest scale systems, often less than $1m^3$, contain either a natural species assemblage to assess effects in the laboratory or in the field, or an 'artificial' community for testing in the laboratory. However, the size and limited biological complexity in these studies with small model ecosystems often limit their ability to predict effects in the real world, especially regarding functional and indirect effects.

As the size of model ecosystems increases there is a greater tendency for them to be outdoors; e.g., ponds and limnocorrals (enclosures within lakes). These outdoor systems range from 1 m^3 to over 1,000 m^3 and have been constructed in variety of ways. They usually contain natural sediment and some include a littoral compartment incorporating aquatic macrophytes. The larger systems are generally more difficult to replicate but overcome many problems of scale and limited biological complexity that small systems may present. In addition, they are often able to sustain studies for several weeks or months and can therefore better assess both the direct and indirect impact of toxicants on a community and, if required, recovery after treatment. Static freshwater model ecosystems have been most widely used to examine the fate and effects of pesticides, particularly effects following single or repeated discrete applications.

Figure 2 schematically summarises the relationship between size (x-axis) and test location, duration, type of community and biological complexity. The width of the bars represents the relative number of studies published.

A more-detailed description of static freshwater model ecosystems that have been used to study the effects of chemicals is given in Appendix A.1.

Figure 2: Relationship between Volume and Several Descriptors for Static Freshwater Model Ecosystems



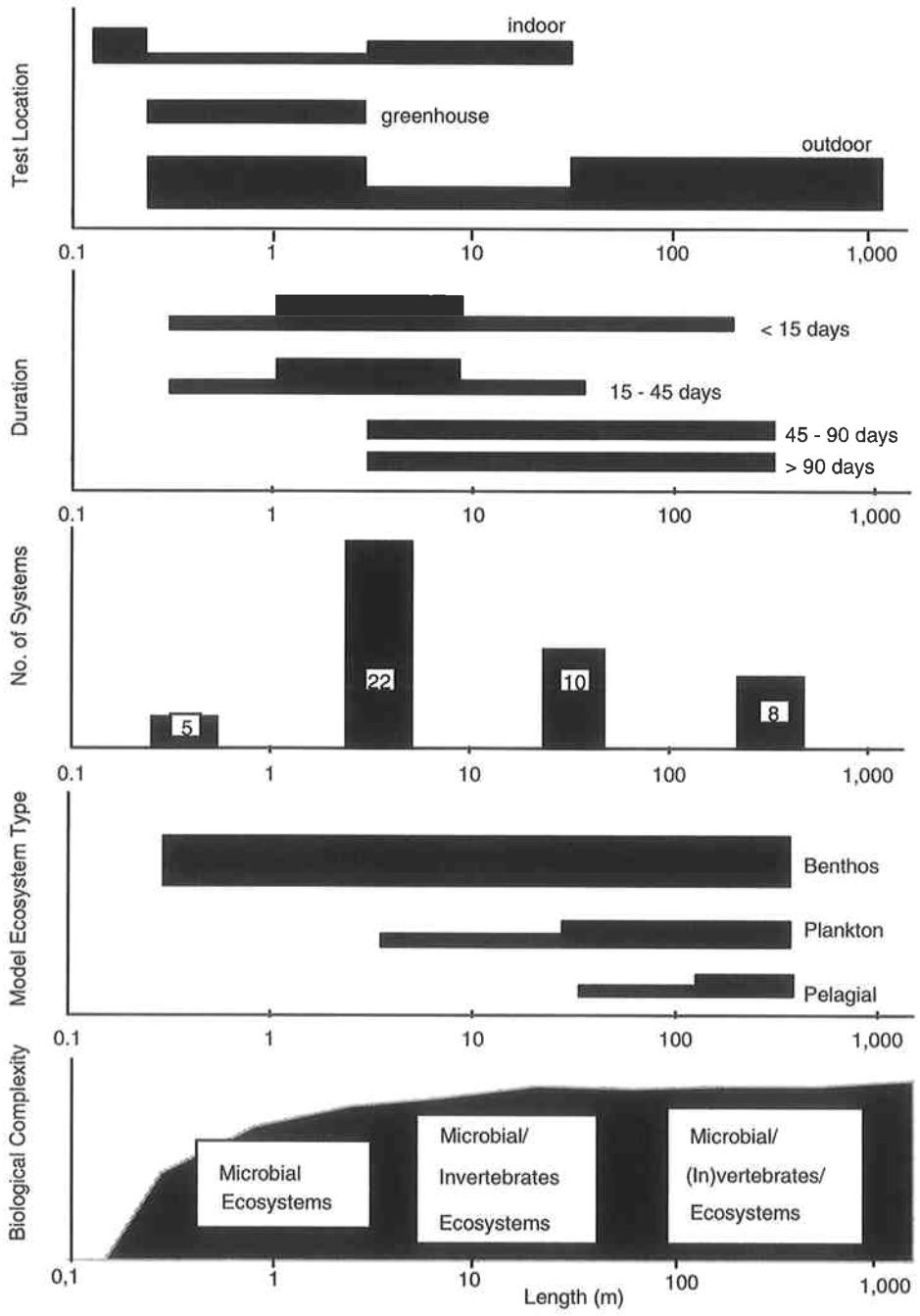
3.3 FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Various flowing freshwater model ecosystems have been developed for use in ecotoxicology including re-circulating throughs, once-through channels, in-stream flumes and large outdoor systems. Test systems described in the literature range from 0.33 to over 500 m in length (Figure 3). Typically, great care has been taken to develop complex and realistic benthic assemblages for testing. The size of the test system dictates the number of trophic levels included and whether the communities include plankton and pelagic (i.e., fish) species. Highly complex microbial ecosystems are attainable in systems <1m in size. Invertebrate communities are often the focus of flowing freshwater studies with emphasis on sensitive taxa such as mayflies, stoneflies and caddisflies. Most test systems have been between 1 and 10 m in length; however, some of the more-striking examples of comparative model ecosystem ecotoxicology are with smaller (2 m in size or less) systems. The duration of tests covers a wide range depending on the objectives of the investigation. Small systems (< 1 m) which appear to be less internally sustainable, are often assessed for less than one month. Study length increases as system size and the ability to maintain ever more-complex communities increases. Not surprisingly, small systems are primarily used under indoor laboratory conditions whereas large systems tend to be outdoor.

Figure 3 schematically summarises the relationship between size (x-axis) and test location, duration, type of community and biological complexity. The width of the bars represents the number of studies published.

A detailed summary is given in Appendix A.2. A more-detailed analysis and the corresponding literature are provided in Appendix B.

Figure 3: Relationship between Length and Several Descriptors for Flowing Freshwater Model Ecosystems



3.4 MARINE MODEL ECOSYSTEM STUDIES

Marine model ecosystems can be conveniently divided into indoor and outdoor systems.

Indoor systems are usually small flow-through systems, often designed to study sediment-water interactions of the microbial, benthic macro-invertebrate or periphyton communities. Consequently, the average size of indoor systems is between 0.01 and 0.2 m^3. The duration of tests in indoor systems is variable and depends on the purpose of the studies.

Outdoor systems are generally larger than indoor systems. The typical size range is between 1 and 150 m^3 with a few systems as large as 1,400 m^3. Outdoor systems have frequently been used to study effects on plankton communities and in some cases on the broader pelagic system. The duration of the tests ranges from a few days up to several months and is related to the size and the complexity of the systems (Appendix A.3).

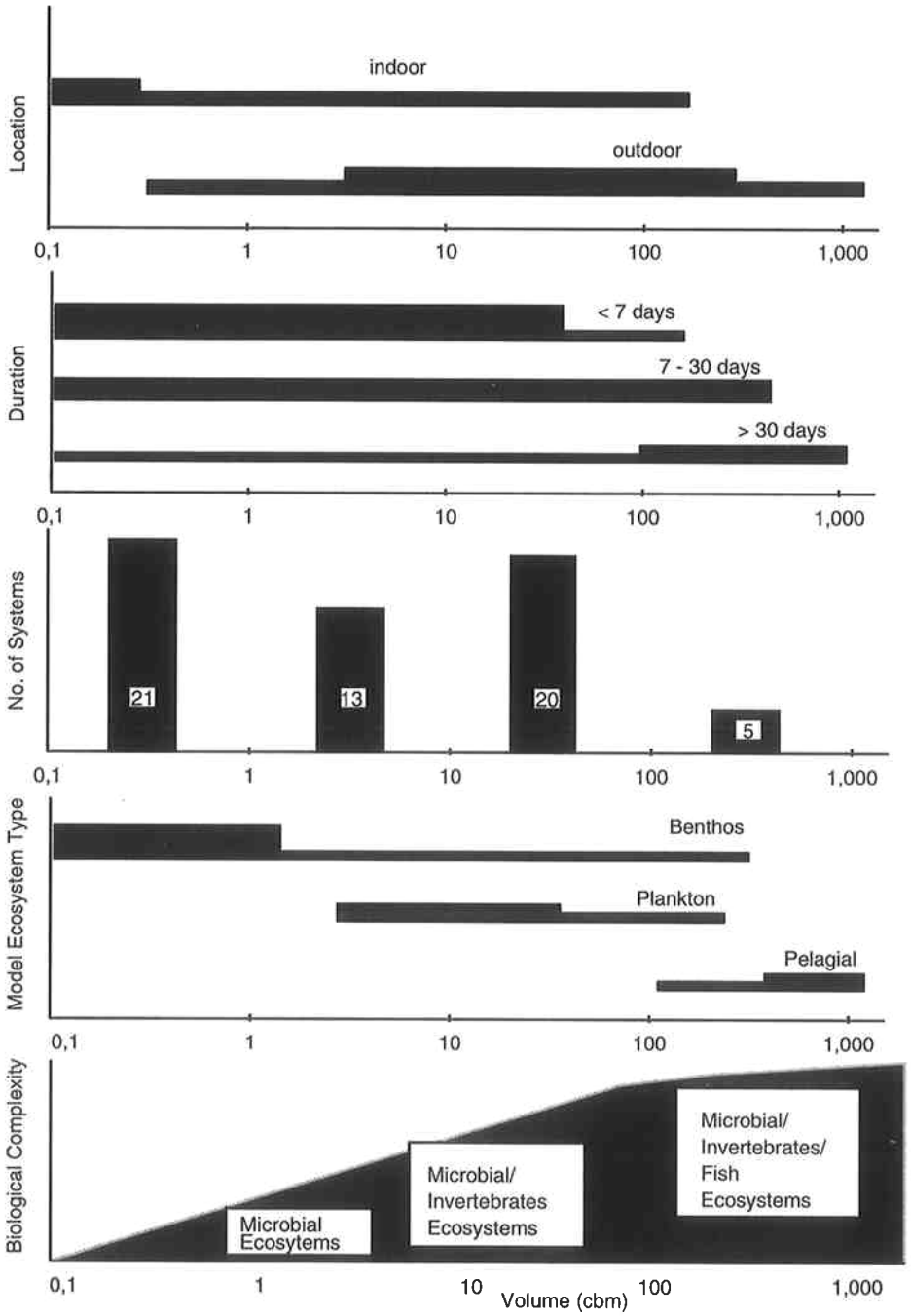
Two basic types of marine outdoor model ecosystems are described in the literature:

- **constructed outdoor** systems which typically consist of moored tanks that contain water, organisms and in some cases a sediment layer. Usually unfiltered sea water is pumped continuously into the microcosms.

- ***in situ* enclosures** are frequently floating plastic bags enclosing a portion of the water column. Optimum dimensions of enclosures have been suggested for ecotoxicological experiments, excluding fish and other large carnivores, to be about 2 to 10 m^3 (depending on the degree of oligotrophy of the system). The optimal duration of the tests is recommended to be less than 4 to 6 weeks, because with time the plankton community diverges more and more from the natural situation.

In marine model ecosystems, as with others, the degree of similarity with the natural environment and the stability of the community tends to increase with the dimensions of the system. Figure 4 summarises some key aspects of marine model ecosystems that are described in the literature.

Figure 4: Relationship between Volume and Several Descriptors for Marine Model Ecosystems



The majority of model ecosystem studies with benthic communities have been performed in systems of > 1 m^3 and with plankton communities in systems ranging from 1 to 80 m^3. On a few occasions very large systems have been used to study effects on the whole of the pelagic community (i.e., including higher carnivores). The duration of the tests reported is variable but generally the larger systems have been used for longer periods. Also, the complexity of the biocoenosis generally increases with the volume of the test system, up to a volume of about 10 to 20 m^3. Thereafter, the complexity only increases when the systems are large enough to support higher carnivores (i.e., > 1,000 m^3).

Marine model ecosystems can help to characterise the effects of substances in the marine environment if the studies are focused on key structural or functional endpoints. However, well-designed model ecosystem studies can be very costly and have only been used by relatively few laboratories located near a natural marine environment. Their use is therefore likely to be limited to special situations that cannot be evaluated in conventional laboratory studies or more cost-effective model ecosystem studies.

4. RESULTS

4.1 CHRONIC SINGLE-SPECIES TO MODEL ECOSYSTEM EXTRAPOLATIONS

To establish the potential usefulness and role of model ecosystems in risk assessment, it is necessary to compare the sensitivity of NOECs obtained from well-designed model ecosystem studies with NOECs obtained from laboratory single-species chronic toxicity tests on the one hand and to field monitoring studies on the other. These comparisons which are performed in this and the following section will provide information on possible extrapolations from single-species tests to model ecosystems and to the real world. They will also provide information to what extent the results of model ecosystem studies can be used to refine a PNEC derived by applying an assessment factor to the results of single-species tests.

4.1.1 Description of the Database

To compare the effects observed in model ecosystem studies with those from single-species tests, only those endpoints where a NOEC and a LOEC value were given, or could be derived from the publication, were used. Through this selection a total number of 248 entries of the original 1108 entries could be used. These data covered for 34 chemicals (Fig. 1). The majority of these were from flowing freshwater systems (Table 1).

Table 1: Number of Chemicals for which Model Ecosystem NOECs and LOECs were available

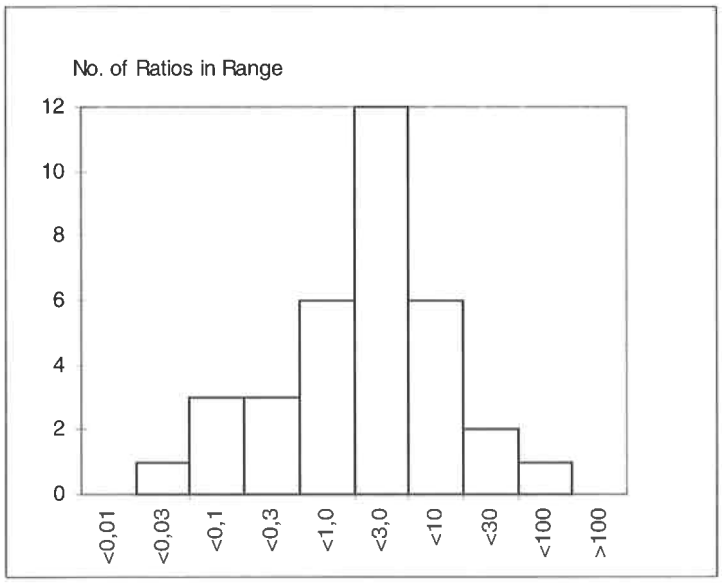
All Test Systems	Marine	Static Fresh-water	Flowing Fresh-water
34	3	5	26

For these 34 chemicals, chronic single-species NOECs for fish, invertebrates and algae were extracted from the EAT database (ECETOC, 1993b), the Pesticide Manual (Tomlin, 1994), company information or from peer reviewed publications. Where no chronic data were available, acute data divided by 10 were taken instead. This is indicated in Appendix C in the column "Remarks" for each individual case.

4.1.2 Comparison of NOECs from Single-species and Model Ecosystem Tests

The 34 single-species NOECs included 17 substances for which fish, invertebrate and algal single-species NOECs were available. Another 14 had at least two single-species NOECs (Table 2). The ratios of single-species NOECs to model ecosystem NOECs were then calculated and assigned to ranges (Table 3). Appendix C gives all the details including the ratios of lowest single-species chronic NOEC to the lowest NOEC from a model ecosystem test on the same substance. Combining the ratios for all chemical classes suggests a log-normal distribution (see Fig. 5). Whilst the number of ratios for substances tested in specific types of test system (i.e., marine, static or flowing freshwater) is too small to draw clear conclusions (Table 4) it appears that there is a log-normal distribution in all systems. In the following discussion different types of model ecosystem are therefore considered together.

Figure 5: Ratios of Single-species NOECs to Model Ecosystem NOECs for all Chemical Classes



The median ratio of single-species NOEC : model ecosystem NOEC for all 34 chemicals was 1.45 with a 90th percentile of 8.14. This indicates that the most sensitive single-species NOEC is not more than 8.14 times less sensitive than the most-sensitive endpoint observed in a model ecosystem study for 90% of the cases. This suggests that the assessment factor of 10 applied in the EU, and in a somewhat different way in the US, risk assessment schemes to the lowest single-species NOEC (of fish, *Daphnia* and algae) to estimate the PNEC appears reasonable, assuming that the model ecosystems have a sensitivity to chemicals similar to that of natural ecosystems. The 90th percentile ratio of 8.14 is also comparable to the assessment factor of five proposed by ECETOC (1993a) which was derived from a review of 13 industrial and agricultural chemicals.

Table 2: Number of Substances with Single-species NOECs available

	Taxonomic Groups							Total
	Fish + Inverts + Algae	Fish + Inverts	Fish + Algae	Inverts + Algae	Fish only	Inverts only	Algae only	
all NOECs measured	13	8	1	1	0	1	1	25
at least one NOEC calculated	4	4	0	0	0	1	0	9
Total	17	12	1	1	0	2	1	34

If the single-species NOEC: model ecosystem NOEC ratios for the pesticides are compared to those for the other chemicals it is apparent that, whilst there is little difference in the 90th percentiles of the ratios, the median for pesticides is lower than the median for the non-pesticides (0.75 and 1.76, respectively). The range of the ratios for the pesticides covers more than three orders of magnitude but the range for the non-pesticides is two orders of magnitude. The distribution of the ratios around the equivalence point is also different. 54% of the pesticides have a ratio less than 1 (i.e., the sensitivity of the single-species test was higher) but only 29% of the non-pesticides fall into this category. This may reflect differences in the fate of the two groups of substances in the aquatic environment. For example, some species may not be consistently exposed to poorly soluble pesticides in the environment because of partitioning processes whereas in a laboratory test continuous exposure is likely to be more easily achieved. It may also be due to the fact that in the course of the development of pesticides numerous possible specific target organisms as well as non-target organisms which may also be impaired are usually tested individually for their response. Therefore the likelihood of getting a more-sensitive response from a wide array of additional species in a multi-species test is rather small.

Table 3: Ratio of Most Sensitive Single-species NOEC to Most Sensitive Model Ecosystem NOEC (Analysed by Chemical Class)

Range	General organic	General inorganic	Metals	Surfactants	**Non-Pesticides**	Insecticides	Herbicides	Fungicides	**Pesticides**	**Total**
<0.01	0	0	0	0	**0**	0	0	0	**0**	**0**
0.01-0.03	0	0	0	0	**0**	0	1	0	**1**	**1**
0.03-0.1	0	0	0	0	**0**	1	1	1	**3**	**3**
0.1-0.3	1	0	0	1	**2**	1	0	0	**1**	**3**
0.3-1	1	1	1	1	**4**	0	2	0	**2**	**6**
1-3	0	1	2	5	**8**	3	1	0	**4**	**12**
3-10	3	0	1	2	**6**	0	0	0	**0**	**6**
10-30	0	0	1	0	**1**	1	0	0	**1**	**2**
30-100	0	0	0	0	**0**	0	1	0	**1**	**1**
>100	0	0	0	0	**0**	0	0	0	**0**	**0**
TOTAL	**5**	**2**	**5**	**9**	**21**	**6**	**6**	**1**	**13**	**34**
Average	2.36	1.17	7.05	2.27	**3.33**	2.70	13.53	0.10	**7.50**	**4.92**
Std.Dev.	1.862	0.837	9.475	1.907	**4.981**	4.16	31.35	--	**21.23**	**13.54**
Median	3.20	1.17	1.50	1.85	**1.76**	1.35	0.71	0.10	**0.75**	**1.45**
90%-percentile	4.14	1.64	17.31	4.93	**6.13**	6.65	39.85	0.10	**9.26**	**8.14**
Max. Value	4.55	1.76	22.86	6.13	**22.86**	11.00	77.5	0.10	**77.50**	**77.50**
Min. Value	0.200	0.578	0.520	0.278	**0.200**	0.080	0.020	0.098	**0.020**	**0.02**

Consideration of the ends of the range of the data set containing the single-species NOEC: model ecosystem NOEC ratios (ratios <1.0 and ratios >10) of all the substances provides a useful comparison with the assessment factors used in risk assessment schemes.

13 of the 34 ratios (38%) are less than 1. There are three probable explanations for an experimental ecosystem test to give a less-sensitive response to a substance than a chronic laboratory test with fish, *Daphnia* or algae.

Table 4: Ratio of the Most Sensitive Single-species NOEC to the Most Sensitive Model Ecosystem NOEC (Analysed by Test System)

Range	All Test Systems	Marine	Static Fresh-water	Flowing Fresh-water
<0.01	0	0	0	0
0.01-<0.03	1	0	1	0
0.03-<0.1	3	0	0	3
0.1-<0.3	3	0	1	2
0.3-<1	6	0	2	4
1-<3	12	0	1	11
3-<10	6	3	0	3
10-<30	2	0	0	2
30-<100	1	0	0	1
≥100	0	0	0	0
TOTAL	34	3	5	26
Average	4.92	3.65	0.61	5.90
Std. Dev.	13.54	0.78	0.59	15.39
Median	1.45	3.20	0.66	1.55
90%-percentile	8.14	4.28	1.20	10.00
Max. Value	77.50	4.55	1.50	77.50
Min. Value	0.020	3.200	0.020	0.050

1. The most sensitive taxonomic group tested as a single-species was not present in the experimental ecosystem test.
2. The substance tested was less bioavailable in the model ecosystem test than in the single-species tests. This means that although the measured concentration of the test substance may have been similar in both single-species and model ecosystem tests, a smaller fraction of the substance in the model ecosystem test was available in a toxic form than in the respective single-species tests. This is possible since the model ecosystem tests are likely to have been conducted with natural water containing dissolved organic substances and suspended solids that might associate with the test substance thereby reducing its toxic effects, as in the case of several heavy metals, for example. The single-species tests, however, are more likely to have been conducted in a water with a lower capacity to complex test substances.
3. Laboratory species are more sensitive than the environmental species for the given material.

In this database all these explanations are likely to apply although more than half of the ratios <1 represent tests in which the most-sensitive taxonomic group in the single-species test was also present in the model ecosystem test, suggesting that bioavailability and sensitivity differences were more likely to be responsible factors.

The potential for a given concentration of a substance to evoke reduced toxic effects under more natural test conditions is an important reason for using results of model ecosystem tests in risk assessment, particularly when the single-species NOECs indicate the possibility of effects.

This report develops a generic assessment factor based on single-species chronic NOEC: model ecosystem NOEC ratios. This conservative approach used the most sensitive single-species endpoint and the lowest NOEC generated for *any* endpoint in the model ecosystem for the same substance. The model ecosystem NOEC, as used above is not necessarily ecologically relevant and the effect observed may not even be an *adverse* effect. In order to best utilise model ecosystem studies in risk assessment the design, interpretation and ultimate conclusions drawn from them must be considered on a case-by-case basis. This conclusion has been drawn by, and reinforced in, numerous scientific fora involving regulators, academia and the private sector (SETAC Europe, 1991; SETAC-RESOLVE, 1992; Graney *et al*, 1994; Hill *et al*, 1994).

The following three model ecosystem assessments of dodecyl alkyl sulphate, copper, and terbuthylazine are taken as examples to demonstrate the use of the conservative generic assessment factor approach in comparison with the case-by-case evaluations of results.

Alkyl Sulphate (AS)

Dodecyl sulphate was evaluated in a 12 m model stream ecosystem in a once-through exposure design receiving river water. Exposures were carried out for 56 d at 6 concentrations. The lowest NOEC in the model ecosystem test was for auto- and heterotrophic microbial lipid class partitioning at 0.02 mg/L. During exposure the bacterial community underwent a structural change (measured using lipid profiles) as certain populations used AS as an energy substrate. The NOEC value of 0.02 mg/L is not the most ecologically relevant to assess toxicity because the alterations in lipid profiles are a direct result of acclimation to AS biodegradation (Guckert *et al*, 1996) which should be considered an environmentally desirable response. Belanger *et al* (1995b) and Guckert *et al* (1996) clearly demonstrated relevant toxic effects to sensitive mayflies and other invertebrates at 0.582 mg/L (LOEC) with a NOEC of 0.224 mg/L based on reduced abundance and biomass. The ecologically relevant NOEC for this model ecosystem test was 0.224 mg/L, and not the 0.02 mg/L NOEC used to develop the generic assessment factor of 8.14 in this report. This NOEC compares favourably with the most-sensitive species, an invertebrate (clam, *Corbicula fluminea*) which had a NOEC of 0.418 mg/L. If the single-species chronic NOEC is divided by 8.14, the result is a PNEC of 0.051 mg/L. Considering the ecologically relevant NOEC of 0.224 mg/L for the model ecosystem study the PNEC derived from the single-species assessment is conservative by a factor of 4.4 (0.224/0.051).

Copper

Copper (Cu) is an essential trace element for living organisms but high concentrations could cause detrimental effects. It was studied extensively in single-species laboratory tests and model ecosystems. The lowest single-species NOEC in the data base is 0.0013 mg/L (ECETOC, 1993b). Assuming only single-species toxicity data were available, a PNEC derived from the application of the single-species: model ecosystem assessment factor (8.14) to this lowest NOEC would be 0.00016 mg/L. Since the assessment factor itself was derived from the lowest of any endpoint measured irrespective of its ecological relevance, its application to a single-species NOEC is expected to give a conservative estimate of the PNEC.

Comparison of the PNEC for Cu with available model ecosystem NOECs indicates that this is the case. The lowest model ecosystem NOEC (0.0025 mg/L) found in the literature was derived from several invertebrate endpoints including drift, abundance, and richness (Leland, 1989). In this case the lowest NOECs are also considered to be ecologically relevant. Considering the NOEC of 0.0025 mg/L for the model ecosystem study, the PNEC derived from the single-species assessment is conservative by a factor of 15.6 (0.0025/0.00016).

Of course, since several model ecosystem studies on Cu are available, they should be considered, on a case-by-case basis in the derivation of the PNEC.

Terbuthylazine

Terbuthylazine (TBA) is a herbicide with a specific mode of action involving inhibition of photosynthesis by blocking electron transport. Several toxicological endpoints were used in a pond mesocosm study including a number of chlorophyta, phytoplankton diversity and primary production (Huber, 1995b). Primary production, an ecologically relevant endpoint, exhibited the lowest NOEC cited in the pond study (0.005 mg/L). The most sensitive single-species NOEC, also algae, was 0.0033 mg. Application of the single-species:model ecosystem assessment factor of 8.14 to the most sensitive single-species data (0.0033/8.14) results in a PNEC of 0.00041 mg/L in the absence of model ecosystem data. The ratio of the predicted PNEC based on model ecosystem versus single-species data (0.005/0.00041) is approximately 12 and suggests again that the single-species extrapolation is indeed conservative when using the single-species: model ecosystem assessment factor.

4.2 MODEL ECOSYSTEM TO FIELD EXTRAPOLATIONS

There is a body of evidence that indicates that well-designed model ecosystems can be considered to represent natural systems sufficiently well. Effects of chemical treatments in the model ecosystems will therefore closely relate to effects likely to be seen in natural ecosystems. The evidence on which this is based includes:

1. direct experimental dosing of natural systems with single chemicals to allow comparisons with single-species and model ecosystem data;
2. simultaneous assessments of single chemicals and effluents dosed into model ecosystems compared with natural systems perturbed and dominated by the same chemical as part of an effluent;
3. comparisons of consistency of results collected from different tests and model ecosystem types for the same chemical; and,
4. evaluation of model ecosystem complexity compared to natural systems.

4.2.1 Direct Experimental Dosing of Natural Systems

The few examples of direct experimental dosing of chemicals into natural systems show that natural systems are as, or less, easily perturbed by chemicals than model ecosystems.

Geckler *et al* (1976) dosed a small calcareous stream (hardness as $CaCO_3$ ranged from 226-310 mg/L) with 0.12 mg/L Cu for 33 months. The stream possessed a diverse flora and fauna of 34 fish species, 109 invertebrate taxa (both from Geckler *et al* 1976), and approximately 100 algal taxa (Weber and McFarland, 1981). Fish populations survived and reproduced in stream reaches when copper was < 0.035 mg/L. Sensitive insect taxa were found at control abundances in reaches with 0.017 to 0.058 mg/L. Algal community composition followed a similar pattern. A large battery of single-species acute and chronic tests in laboratory and site waters showed a high degree of correspondence (within a factor of 2) between the field experiment and laboratory toxicity test results.

Lewis *et al* (1986) and Woltering and Bishop (1989) summarised the effects of dodecyl trimethyl ammonium chloride (DTMAC) exposure on a small stream. As with the above copper study, tests were combined with extensive single-species and lentic and lotic model ecosystem assessments (Belanger, 1994). The test material was found to be equally toxic to algae and invertebrates in single-species tests and significantly less toxic to fish; therefore, the stream dosing studies concentrated on the former. DTMAC was dosed in a stream for 3 weeks 1,000 m below a wastewater treatment plant at 0.25 mg/L. Fate and effects were followed for 3,760 m downstream as DTMAC was biodegraded.

There were no effects on periphyton, macroinvertebrates and fish when concentrations were at or below 0.25 mg/L in the stream. *Daphnia magna* held in cages at the stream site for 7 days had reduced survival at concentrations greater than or equal to 0.115 mg/L. These results are within a factor of 2 of NOECs (NOEC range of 0.185-0.234 mg/L) derived from model stream ecosystem studies summarised by Belanger (1992, 1994).

Borthwick *et al* (1985) and Clark *et al* (1987) compared laboratory-generated toxicity data with responses of estuarine shrimp and sheepshead minnows exposed in areas oversprayed by the insecticide fenthion. Field effects were predicted by laboratory toxicity tests if the applications in the laboratory were "pulsed" as in the field. Field observations of mortality of >40% occurred when the laboratory LC_{50} was exceeded.

4.2.2 Model Ecosystem Studies of Effluents and Single Chemical Impacts

The responses of model ecosystems and communities to effluents under field conditions has been compared in a few studies. Niederlehner *et al* (1990) and Pontasch and Cairns (1991) simultaneously assessed the response of protozoans on artificial substrates in static 7.5L systems and macroinvertebrates in 1.7m recirculating model streams with protozoans and invertebrates in a receiving stream exposed to a complex effluent. The effluent constituents included chlorides, ammonia, phenols and lead. Laboratory model ecosystems were affected at similar concentrations to those that caused effects in the field. NOECs for protozoan community structure ranged from 0.3 to

1% effluent whereas for invertebrate community structure the NOEC was 0.1 to 1% effluent. Field observations of protozoan and invertebrate population and community effects were consistent with these observations, with field NOECs in the range of 1.1 to 4.1% effluent. The lower NOEC from the invertebrate model ecosystem study should be cautiously interpreted as a dilution factor of 10 was used in the study (100, 10, 1, 0.1% effluent plus control treatments), whereas a dilution factor of 3 was used in the protozoan studies.

An extensive assessment of the impacts of copper from cooling tower blowdown on the Clinch River, Virginia, was made by Farris *et al* (1988; 1991), Clements *et al* (1989) and Belanger *et al* (1990). This included 16 single-species laboratory chronic tests, model stream ecosystem studies with cooling tower and site water, caged *in situ* bioassays, and invertebrate field surveys over a four-year period. Information derived from model stream ecosystems resulted in a NOEC of <0.012 mg/L, but site water (upstream control) consistently contained some 0.001-0.005 mg/L of copper. Field responses were entirely consistent with model ecosystem studies. At sites downstream of the copper-dominated effluent full recovery did not occur until copper concentrations were <0.012 mg/L. Single-species tests with endemic molluscs, insects and fish also showed the most sensitive species (all unionid and corbiculiid bivalves) had chronic NOECs of 0.012 mg/L. Caged clams exposed in the river were affected when concentrations of copper exceeded 0.015 mg/L.

4.2.3 Comparison of Results from Different Model Ecosystems

Confidence in the effects assessment of a chemical should increase if results from different model ecosystems provide the same or similar result. This would be most likely if the test systems were each mimicking fundamental properties of natural systems. On occasion researchers have attempted to corroborate results by conducting tests with the same chemical under different conditions.

Pratt *et al* (1989) evaluated protistan community responses to phenol using the same experimental system design in two different locations (Kentucky and Virginia, USA). The model ecosystem NOECs were 3.0 and 1.2 mg/L, respectively, based on measured concentrations with the nominal concentrations both being 3.0 mg/L. LOECs were 11 and 7.7 mg/L, respectively. The results were equivalent. Differences in the final NOEC were mostly due to the choice of test concentrations.

In a study of selenium toxicity, Pratt and Bowers (1990) evaluated communities in model ecosystems under laboratory conditions and compared them to protistan community responses in a large model stream. They concluded that responses under the two very different conditions were similar. Using the same model ecosystem design, Clements *et al* (1989) determined the responses of aquatic macroinvertebrates to copper at two different field sites. Invertebrates were more affected by copper

in softer water. NOECs were <0.006 and <0.012 mg Cu/L at hardnesses of 60 and 153 mg/L, respectively. The slope of the exposure-response curve was similar under both conditions.

Belanger (1992) determined the effects of dodecyl trimethyl ammonium chloride (DTMAC) on model stream ecosystems. NOECs in summer/fall and winter/spring conditions were 0.235 and 0.185 mg/L, respectively. Woltering and Bishop (1989) and Lewis *et al* (1986) reported a series of NOECs for the effects of DTMAC on periphyton communities developed on glass slides and exposed in the field and in an indoor stream. NOECs ranged from 0.110-0.250 mg/L, entirely consistent with the findings of Belanger (1992).

4.2.4 Model Ecosystem and Natural Ecosystem Complexity

Comparison of the biological complexity of model ecosystems to that of natural systems provides an indication of how well experiments in model ecosystems might predict responses in natural systems to exposure to chemicals. For example, Pontasch and Cairns (1991) found 35 macroinvertebrate taxa present in the model stream ecosystems compared to 27 in the natural system being evaluated. Belanger *et al* (1995a) and Cuffney *et al* (1990) showed invertebrate community similarity, as measured by Jaccard's coefficient of community or Stander's SIMI, exceeded 0.7 for model stream ecosystems in southwest Ohio compared to the source river from which the test facility drew water.

Some researchers have quantified the degree to which variance in biological complexity can be controlled in model ecosystem studies and under field conditions. Pratt and Bowers (1990) summarised over one dozen community metrics used in small model ecosystem studies and found coefficients of variation in laboratory model ecosystems ranging from 5-30%. The same range (5-30%) approximates coefficients of variation for field based studies using the same metrics. Guhl (1994) presented a summary of 10 years of research on aquatic staircase model ecosystems (a flow through model) in comparison to aquatic assemblages in the field (Guhl, 1987; Scholz and Müller, 1992). Comparisons of algae, protozoan, and small metazoan species present in the model ecosystems with those of the upper and lower River Rhine showed 69% and 86% community similarity, respectively. Investigations of successional changes and the influence of river discharge in both the field and model ecosystems demonstrated a high degree of similarity. Diatom maxima in late fall and end of winter were observed in both situations (Mauch, 1988). When model ecosystems were constructed to simulate the flow rate in the river, the community compositions in the field and model systems were comparable.

5. CONCLUSIONS AND RECOMMENDATIONS

1. There is considerable experience in the use of static and flowing freshwater model ecosystems, and to a lesser extent of marine systems, to assess the impact of a wide variety of chemicals.
2. It is not desirable to recommend a standard, or set of standard, test procedures for model ecosystem studies; each study should be designed to address the specific issues/data requirements that have arisen from earlier stages of testing.
3. A comparison of the most sensitive NOECs from single-species chronic toxicity tests with those from model ecosystem studies for 34 chemicals gave a median ratio of 1.45 with a 90th percentile of 8.14. This indicates that an assessment factor of about eight used on the most sensitive single-species NOEC would be protective of the most sensitive endpoints in model ecosystem studies for 90 percent of cases. This value is in close agreement to the assessment factor of five previously proposed by ECETOC (1993a).
4. For the majority of chemicals and effluents reliable risk assessments can be carried out without the need for model ecosystem studies, assuming that sufficient data exist from single-species tests. However, model ecosystem studies may provide valuable additional information if the PEC/PNEC ratio approaches or exceeds unity, particularly if the physico-chemical properties of the chemical suggest the possibility of reduced bioavailability in the real world situation.
5. The evidence from experiments indicates that well-designed model ecosystems can represent many of the most important features of real-world ecosystems. These experiments include studies in natural systems, effluent studies, comparison of the biological complexity of model ecosystems with that of natural systems, and studies of different model ecosystems dosed with the same toxicant. Additionally the results indicate model ecosystem NOECs to be similar to NOECs determined from experiments in natural systems.
6. Given that effects in well-designed model ecosystem studies closely relate to those seen in natural ecosystems, assessment factors for NOECs from such studies to "real-world" ecosystems should in general be one, or close to one.

APPENDIX A. DESCRIPTION OF TEST SYSTEMS

A.1 STATIC FRESHWATER SYSTEMS

This chapter gives an overview of the types of system that are used to study biocoenoses with emphasis on those that have been used to study the effects of chemical stressors. It is not intended to give a comprehensive review of experimental lentic freshwater multispecies tests because there are several recent reviews which provide a wealth of information about these systems (SETAC Europe, 1991; Crossland *et al*, 1992a; Cairns and Cherry, 1993; Rosenberg and Resh, 1993; Graney *et al*, 1994).

Man-made systems range from small simple indoor tanks to large-scale outdoor ponds that are arguably indistinguishable from natural ponds. Natural whole lakes have also been used by a few investigators. Some of the advantages and disadvantages of the general types of system are discussed below along with an overview of the different designs.

A.1.1 Small Microcosms

Two approaches have been used to study multispecies effects in small microcosms that attempt to represent specific compartments of static freshwater. The first method is to contain a natural assemblage of organisms in a small volume and assess effects in the laboratory or in the field (Clements *et al*, 1989; Stay *et al*, 1989). This procedure can be relatively well standardised (Leffler, 1981). Field deployed microcosms can be used to study the short-term behaviour of populations in a local area but have limited value in indicating changes in communities in whole ecosystems (Barton and Smith, 1984). The major limitations of such microcosms is that only certain aspects of the aquatic ecosystem are represented. For example, key species may be absent and environmental conditions may quickly diverge from those of a natural environment.

The second approach uses artificial (gnotobiotic) assemblages in the laboratory. This approach has the same types of limitations as microcosms with natural assemblages but differ in that they have been designed to achieve a higher level of biological control. These microcosms have been developed over many years and have been reported to be suitable for standardisation. They typically consist of a relatively-small test vessel, for example containing 3 litres of chemically-defined medium and sometimes sediment. The organisms are selected and introduced as a standard species list. The "Taub microcosm" includes 10 algal and 5 animal species to represent primary and secondary procedures and primary consumers. The small scale of the test vessels and the simple community structure allow for a high degree of replication, typically 6 at each of 4 treatment levels. Each control vessel is expected to follow a predictable biological succession so that both direct and indirect effects can be observed in

treated vessels. The reproducibility and repeatability of the standardised aquatic microcosm has been reported as being good for a range of chemicals (Taub *et al*, 1986).

A.1.2 Small Indoor Mesocosms and Larger Microcosms

These systems can be considered to be more complex, less standardised and somewhat larger in scale than the Taub type microcosm. They include a wide range of designs still with emphasis on a high degree of control and similarity whilst aiming to model more closely specific aspects of static freshwater ecosystems. Typical examples are small (< 1 m^3 and often < 100 litres) indoor vessels, usually glass, occasionally containing sediment. The vessels are often filled with lake or pond water so that suitable organisms are also introduced. Additional introduction of specific groups of invertebrates is often involved. However, because of the limited volume the biological complexity is usually limited to phytoplankton, zooplankton and benthic invertebrates if sediment is included. Some facilities have all of the test vessels interconnected and circulate water through all of them during a colonisation period so that biological development occurs similarly in all. Fish are not usually included because of the inadequate size of the systems. A small number of designs have a sloping sediment leading to a terrestrial compartment (Cole and Metcalf, 1980). These systems have been designed primarily to investigate chemical transport through food chains rather than to model components of a natural ecosystem and consequently have not been used to study wider biological interactions at the community level.

The limited realism of these indoor systems can be considered as a disadvantage compared to more complex systems. For example, some environmental factors such as weather, mixing or thermal stratification can not be simulated, and water chemistry processes may not accurately represent a large natural system. Edge effects may have a significant influence if the ratio of water volume to wall surface area is not sufficiently large. Species richness and abundance are more likely to differ from a natural ecosystem in small systems than in those of larger scale.

Indoor systems can be much cheaper to build and manage and may more easily achieve control of abiotic and biotic factors. This is particularly useful for experiments with statistical designs requiring a high number of replicates. Despite the limited complexity and realism of such designs they have been used successfully in assessing the fate and direct effects of agricultural chemicals (Dortland, 1980). The larger and more-complex examples have also demonstrated indirect effects such as predator - prey and competition interactions.

A.1.3 Outdoor Microcosms

Field mesocosms have often been used to study basic ecological questions such as species succession and nutrient cycling however they are becoming widely used to assess the effects of chemical stressors. There are 2 basic types of system, enclosures and experimental ponds. It is notable that herbicides and insecticides have been the most frequently tested chemicals in both types of system (SETAC, 1991; Crossland *et al*, 1992a; Cairns and Cherry, 1993; Rosenberg and Resh, 1993; Graney *et al*, 1994). To a large extent this reflects the requirements for ecosystem effects data in the regulation of pesticides.

Enclosures

This category represents a wide range of devices that are designed to enclose specific compartments of natural ponds or lakes. The simplest are small-scale bags or cylinders that isolate a relatively small number of species in a limited habitat. For example clear polythene bags have been filled *in situ* with pond or lake water containing phytoplankton in order to monitor the effects of chemicals on photosynthesis under natural conditions. Although these experiments will sometimes have little predictive value other than for the compartment studied, they can be useful in comparisons with laboratory tests or more-complex field studies.

Larger-scale bag enclosures for example, 1 m diameter, 2.5 m deep, have been used successfully to assess the effect of agricultural chemicals on phytoplankton communities in lakes (Yasuno *et al,* 1988; Havens, 1994). Experimental designs have included both floating bags and sealed bags that are suspended in the water column from floating buoys. Studies have varied from short-term experiments lasting a few days to assess the rapid response of phytoplankton to chemical stress, to long-term studies that include a pre-treatment period, an exposure phase and a subsequent period to monitor recovery that may take several months.

One of the main disadvantages of these bags appears to be their low physical strength which may not withstand wave action and can not support a sediment layer. They have the clear advantage of being relatively inexpensive compared to other replicated mesocosms.

The most widely used enclosure design is the limnocorral. These are usually cylindrical structures (although other shapes have been used) that isolate a natural assemblage of organisms between the water surface and the sediment. There are a range of materials used to achieve this purpose. The more frequently used are (i) rigid fibreglass or plastic tubes that can be imbedded into the sediment and stabilised by tethers and (ii) plastic or vinyl liners that are fixed to the sediment and maintained in position by various floating devices. As is often done in smaller scale test systems, limnocorrals are occasionally interconnected to promote similar biological content and complexity in all replicates before treatment.

Limnocorrals cover a great range of volumes, surface areas and depths. Small enclosures may contain 1 m^3 of water in the littoral zone of large (deep) lakes or of shallow lakes or ponds. The largest contain in excess of 100 m^3. These larger limnocorrals are often used for the more complex long-term experiments which frequently include assessments of the impact of chemicals and subsequent recovery. Such large systems can support the inclusion of caged organisms such as fish, to assess the effects on single-species without significantly influencing other aspects of the study. Free-swimming (uncaged) fish have been introduced and have bred in limnocorrals with volumes of about 40 m^3 or less (Siefert *et al*, 1989). However, some authors suggest that much larger volumes are required (Uhlmann, 1985) or simply that fish reproduction can not be adequately studied even in large scale mesocosms (Crossland *et al* 1992a).

There are a few examples of enclosures that include a section of the littoral environment as well as the water column and sediment (Siefert *et al*, 1989; Giddings, 1986). These facilities are designed to investigate effects of chemicals in environments normally dominated by emergent aquatic plants. Because of the relatively-high surface area of the biomass, the bioavailability of chemicals and their distribution in different micro environments are often quite different to that in open water (Siefert *et al*, 1989). This is particularly relevant for chemicals with high adsorptive properties such as some pesticides.

The number of replicates and treatment levels used in limnocorral studies is highly variable reflecting in part the wide range of statistical designs but also the often limited resources available to manage the study.

Experimental Ponds

As with all types of mesocosms, there is a larger range of structural designs of experimental ponds. They fall into 2 main types, those excavated in the ground and those that are walled.

The shape and dimensions of experimental ponds cover a considerable range. Most are of a regular shape and contain 10 m^3 to 1,000 m^3. The larger ponds have minimal wall effects. Depths of the larger ponds do not usually exceed 2 m and smaller ponds are 1 m deep. The sides of ponds may be vertical or of varying slope. Most artificial systems are either built in concrete or are lined with an inert liner to prevent water loss. In both cases a sediment is usually introduced. Facilities typically include a series of 6 to 12 units that are sometimes interconnected to uniformly distribute organisms and water (Crossland, 1988a; Heimbach *et al*, 1992).

Prior to a study a period of several months may be required to establish a suitable flora and fauna. The length of this colonisation period depends on how the ponds are initially prepared. For example, if they are filled with a natural pond water or tap water, and whether they are inoculated with macroinvertebrates

and insects or allowed to colonise naturally. Typical biological communities in the colonised systems include a wide range of phytoplankton, aquatic plants, zooplankton, benthic macroinvertebrates and emergent insect larvae. In many studies free swimming (uncaged) fish are also introduced however, as for the limnocorrals, there are several reports indicating that inclusion of zooplanktivorous fish in particular requires careful study design to ensure that their impact on the community can be adequately interpreted (Crossland *et al*, 1992a). Piscivorous fish have been introduced in some studies to prevent devastation of the zooplankters population (Macek *et al*, 1972; Deutsch *et al*, 1992).

The realism, biological complexity and scale of most experimental ponds enables a great variety of investigations to be undertaken. In addition to the direct effects of chemicals on individual species it is usual to assess aspects of community function such as total respiration and primary productivity. Indirect effects such as predator - prey and competition are often observed, and since most of the experiments are designed to assess single or intermittent exposure, recovery is also monitored.

In general, the larger outdoor mesocosms overcome many of the limitations of microcosms or smaller mesocosms. For example, they can support a similar biological heterogeneity to that found in the whole ecosystem although this has its own problems of increased inherent variability. This and other key advantages and disadvantages of mesocosms are discussed in several reviews (SETAC, 1991; Crossland *et al*, 1992a; Cairns and Cherry, 1993; Graney *et al*, 1994).

A.1.4 Whole Ponds and Lakes

Natural ponds and lakes have been used to assess the fate and effects of various stressors, mainly acidification and nutrient enrichment (Schindler, 1985). These experiments can last many years in order to gain sufficient understanding of the system to discern the effects of the stressor from the inherent variability of the lake or pond. Paleoecological methods have been used to gain information on the history of lakes because of the long time scales of whole lake change. The inability to apply all but the minimal control on natural system is the key disadvantage of such experiments. Mesocosms have a clear advantage in this respect since they can be built in appropriate locations to model specific types of natural system, and replication and testing of a range of concentrations can give the necessary statistical power to discriminate the magnitude of effects. In general, the pond experiments were good predictors of effects in the natural system. Such comparative information is useful in calibrating the mesocosm.

A.2 FLOWING FRESHWATER SYSTEMS

Several reviews have been published in recent years on the use of flowing freshwater streams in ecotoxicological research. The focus of most reviews has been on physical stream design, experimental design, ecological applications and interpretation of direct or indirect effects. Systematic reviews of ecotoxicological conclusions (NOECs and/or ECs) from flowing freshwater stream studies have not been published.

A.2.1 Test System Designs

A large variety of test system designs have been used ranging from small laboratory re-circulating channels to large-scale outdoor streams. The general categories of flowing freshwater systems, in terms of physical structure, can be categorised as:

- re-circulating troughs;
- once-through channels;
- in-stream flumes;
- large scale outdoor.

The test systems can be qualified in at least four physical dimensions - length, width, depth, and volume. Each dimension has an effect on test system suitability and ease of control. The most familiar dimension, test system length, ranged from 0.33 - 540 m. Other dimensions were often less well described in research reports and may be problematic for summarisation purposes. Other physical factors such as inflow rate and water velocity, have not been included in this review, but are important in considering the climax community obtained in any given system.

A.2.2 Model Ecosystems

The development of the ecosystem found in flowing freshwater models depends on several factors:

- source of biota: the development of communities of bacteria, algae, protozoa and small invertebrates is normally autochthonous. Higher invertebrates and vertebrates are autochthonous in some studies but can be experimentally introduced as well. In these communities introduced organisms would be allochthonous;

- system size: in small laboratory models, ecosystems with small invertebrates are easier to maintain; however, maintenance of communities with several trophic levels including higher invertebrates is possible;

- length of study: development of climax communities containing organisms with relatively-long life histories is possible in larger systems or in smaller systems wherein key biological and physical structure is maintained.

All of these model ecosystems can potentially include several different trophic levels even in small laboratory models. Depending on the represented biological complexity, model ecosystems in flowing freshwater models may be partial to relatively complete representatives for natural environments.

A.2.3 Duration of flowing freshwater studies

Study length was broken down into pre-treatment or colonisation, treatment, and recovery phases. Three types of model ecosystem studies were identified:

- a "typical" colonisation and treatment experiment that may or may not have included a recovery phase. Duration of treatment was usually days to months;

- pulse-exposure experiments where the treatment phase was 1 day or less with a considerable emphasis on post-treatment recovery of the biocoenosis;

- colonisation experiments where barren substrates were placed in the test system at the beginning of the initiation of exposure. The ability of organisms to colonise and maintain populations were often emphasised.

Each experimental approach is a legitimate method to answer specific hypotheses. Ultimately, the prediction of chronic effects at the indicated levels will depend on the study length. The data were probed to determine study lengths for each of the types of studies (typical, pulse, colonisation). Most studies were of the typical type with pre-treatment (28 d median), exposure (30 d median), and recovery (45 d median, ignoring studies in which recovery was not measured). Pulse-exposure studies were rare and were limited to exposures to simulate a brief pesticide application. Colonisation studies were relatively common.

A.2.4 Size of test systems

The dimensions of the test systems described in the literature are summarised in Table A.1. It was felt, however, that the simple univariate statistics of mean, standard deviation (SD) and median incompletely described the types of test systems. A log-frequency distribution was used to obtain an impression of the distribution. Log intervals for length, width, depth, and volume are given in Table A.2. The majority of systems had lengths and widths in intervals 3-5 (0.5-4.0 m long and 0.1-0.8 m wide).

Depth and volume-interval distributions were more spread out than those for width and length. The Monticello Ecological Research Station studies stand out at the tails of the distributions being the physically largest site with approximately 10 chemicals and physical disturbances studied over the past 20 years.

In summary, the "average" system tends to be rather large and, thus, physically capable of possessing complex biological communities. This is not surprising as researchers recognised the need to establish larger physical structures to maintain sufficient model ecosystems for testing.

Table A.1: Average and Median Characteristics for Flowing Freshwater Model Ecosystem Studies Considered in this Report

	Length (m)	Width (m)	Depth (m)	Volume (L)
Summarised by Test System				
n	43	43	38	40
mean	50.13	0.58	0.22	38,242
SD	110.32	0.70	0.20	140,395
median	6.10	0.30	0.16	254
Summarised by Test Chemicals Used Across all Systems				
n	108	108	102	105
mean	72.47	0.59	0.21	88,545
SD	159.93	0.65	0.17	238,754
median	4.20	0.33	0.15	61

Table A.2: Log-frequency Intervals Used to Describe the Distribution of Test System Physical Characteristics

Interval	Length Range (m)	Width Range (m)	Depth Range (m)	Volume Range (L)
1	0-0.25	0-0.05	0-0.01	0-2.5
2	0.25-0.5	0.051-0.10	0.011-0.02	2.5-5.0
3	0.51-1.0	0.111-0.20	0.021-0.04	5.1-10.0
4	1.1-2.0	0.21-0.40	0.041-0.08	10.1-20.0
5	2.1-4.0	0.41-0.80	0.081-0.16	20.1-40.0
6	4.1-8.0	0.81-1.60	0.161-0.32	40.1-80.0
7	8.1-16.0	1.61-3.20	0.321-0.64	80.1-160
8	16.1-32.0	3.21-6.40	0.641-1.28	161-320
9	32.1-64.0		1.281-2.56	321-640
10	64.1-128.0		2.561-5.12	641-1,280
11	128.1-256.0			1,281-2,560
12	256.1-512.0			2,561-5,120
13	512.1-1024			5,121-10,240
14				10,241-20,480
15				20,481-40,960
16				40,961-81,920
17				81,921-163,840
18				> 163,840

Table A.3: Numbers of Test Systems (Flowing Freshwater Model Ecosystem Studies) Found for Each Interval

Interval	Length Interval	Width Interval	Depth Interval	Volume Interval
1	15	0	2	3
2	2	1	0	1
3	21	28	5	1
4	16	45	3	23
5	17	11	55	11
6	4	11	21	14
7	6	10	13	1
8	3	1	3	9
9	11			11
10	1			5
11	2			1
12	12			4
13	10			0
14				5
15				1
16				0
17				3
18				12

A.2.5 Substances Evaluated in Flowing Freshwater Studies

Organic compounds and metals accounted for the majority of studies in flowing freshwater systems. Effluent studies were commonly evaluated, and a distinct goal of many biocoenosis studies was to determine chemical - chemical or chemical - biota interactions. Surfactants, as a class, were studied very frequently. Zinc and copper were most frequently studied of the heavy metals. The herbicide atrazine was among the most studied organics other than surfactants. Studies with different chemical classes were relatively-well distributed amongst the different test systems.

Table A.4: Number of Studies Reported for Various Groups of Substances

Substance Group	Number of Studies Reported			
	1-5	6-10	11-20	>20
Metals				
Aluminium	X			
Cadmium	X			
Chromium	X			
Cobalt	X			
Copper				X
Mercury	X			
Selenium	X			
Zinc			X	
Inorganics				
Chlorine		X		
Nitrate	X			
Phosphate	X			
Ammonia	X			
Organics				
Pesticides				X
Surfactants				X
Other organics				X
Effluents				X
Interactions/Mixtures			X	

Table A.5: Summary of Chemicals Represented in Flowing Freshwater Biocoenosis Studies

Chemical Class	Sub-class	Studies in Sub-class	Studies in Class
Physical Factors	heat	5	
	pH (acid and alkaline)	6	
	sediment	2	
	UV-radiation	1	14
Metals	aluminium	4	
	cadmium	3	
	chromium	3	
	cobalt	1	
	copper	21	
	mercury	1	
	selenium	2	
	zinc	12	47
Inorganics	ammonia	3	
	chlorine	9	
	nitrate	2	
	phosphate	2	
	sodium chlorate	1	17
Organic Compounds	anthracene	3	
	atrazine	8	
	chlorphoxim	1	
	chlorpyrifos	2	
	dextrose	1	
	diazinon	1	
	dichloroanline	1	

Table A.5 (cont.): Summary of Chemicals Represented in Flowing Freshwater Biocoenosis Studies

Chemical Class	Sub-class	Studies in Sub-class	Studies in Class
Organic Compounds (cont.)	diflubenzuron	1	
	fenvalerate	1	
	hexachlorobiphenyl	1	
	lindane	1	
	MSMA	1	
	p-cresol	1	
	paraquat	1	
	pentachlorophenol	1	
	phenol	1	
	sucrose	2	
	surfactants	19	
	temephos	1	
	trifluralin	1	
	trifluoromethylnitrophenol	3	52
Effluents	acid mine drainage	1	
	coal leachate	1	
	complex industrial	2	
	contaminated river water	1	
	JP-4 water soluble	1	
	municipal sewage	2	
	kraft mill effluent	3	
	oil shale refinery	1	
	paper mill effluent	1	
	petrochemical	1	14

Table A.5 (cont.): Summary of Chemicals Represented in Flowing Freshwater Biocoenosis Studies

Chemical Class	Sub-class	Studies in Sub-class	Studies in Class
Interactions	aluminium-pH	2	
	ammonia-chlorine	2	
	cadmium-zinc	1	
	cadmium-copper-zinc	1	
	chlorine-graser intensity	2	
	cobalt-copper-zinc	1	
	copper-manganese-chromium	1	
	copper-zinc	1	
	pesticide-sediment	1	
	pH-zinc	2	
	sewage-surfactants	2	16

Note the sum of all studies exceeds that of the number of flowing freshwater studies cited in Table A.1. Studies involving interactive or multiple stressors may be counted as additional studies if designed as two-factor with interaction investigation.

A.2.6 Ecotoxicological endpoints in flowing freshwater model ecosystem studies

Not only are test design and climax communities different in studies with flowing freshwater model ecosystems, but so are the chosen ecotoxicological endpoints used for assessing effects. Each endpoint can be a legitimate method to answer specific hypotheses. For the most important goal of model ecosystem studies, that of protecting the environment, it is important to know the influence of a substance to the entire community.

Some of the reviewed studies did not show specific data which drives the stated conclusion. In these limited cases, EC_{50} or NOECs may be quoted for selected population, community structure, or community function measurements. Many studies provided holistic assessments of the entire community contained in the model.

A.3 MARINE SYSTEMS

Ecological research in marine micro- and mesocosms started in the early sixties, where ecologists used large plastic enclosures to isolate portions of the pelagial to measure phytoplankton dynamics (McAllister *et al*, 1961; Antia *et al*, 1963). Beside this large *in situ* experiments, other investigators used constructed outdoor tanks to study phyto- and zooplankton dynamics (Strickland and Terhune, 1961). Odum *et al* (1963) used outdoor microcosms containing pelagic and benthic communities to include benthic processes and nutrient cycling. The smallest microcosms cited in the literature of this period are static indoor microcosms to study microbial sediment - water interactions (Abbott, 1966).

In the seventies, ecotoxicologists started to use micro- and mesocosms to test the effects of chronic pollution and eutrophication on marine pelagic systems. Since then, several comprehensive reviews have been published, focusing on the use of marine micro- and mesocosms for fundamental and applied ecological research and on the evaluation of micro- and mesocosms for assessing contaminant effects in marine systems (Grice and Reeve, 1982; Adams and Giddings, 1982; Odum, 1984; Clark and Cripe, 1993; Clark and Noles, 1994). Therefore, the following description of test systems will be limited to some representative test designs, where effect concentrations for a number of chemicals were measured. For more-detailed information on the wide range of other test designs, readers are referred to the reviews cited above.

The experimental test systems can be categorised as indoor and outdoor systems, respectively.

Indoor systems are usually small flow-through systems, designed to study sediment-water interactions of the microbial biocoenosis or benthic macroinvertebrate or periphyton communities. Consequently, the average size of indoor systems is between 0.01 and 0.2 m^3. Only very few systems are of greater dimensions and designed to study the processes in the water column. The duration of tests in indoor systems is very variable and depends on the purpose of the studies (Figure 4).

One of the best-evaluated indoor systems are the benthic tanks of the EPA Environmental Research Laboratory at Gulf Breeze (Tagatz and Ivey, 1981). The authors used sand-filled laboratory boxes of about 30 x 30 cm that were colonised for a period of 8 weeks by settling of planktonic larvae entrained in continuously-supplied unfiltered seawater from the nearby sea. An other test design was to use sand filled boxes located by scuba divers in 3 meter depth for colonisation by naturally occurring animals. Thereafter the boxes were transferred in the laboratory and continuously supplied with unfiltered seawater. Both systems were used to study the reproducibility of the measurements and the effects of a number of chemicals on the benthic macroinvertebrate population. For this purpose, the test chemicals were metered by pump to the incoming sea water to give the desired test

concentrations during the defined exposure period. The perturbation period usually lasted about one to two weeks. Thereafter the benthic cores were sampled and the biological composition evaluated.

A total number of 21 chemicals was tested in the different indoor systems reviewed for this chapter. The major chemical classes tested were:

- general chemicals ~ 40 %
- pesticides ~ 30 %
- heavy metals ~ 10 %

About 20 % of the studies were performed without any perturbator or with variable abiotic factors of ecological relevance.

Outdoor systems generally are of greater dimensions compared to indoor systems. The typical size range is between 1 and 150 m^3. Some few systems have extensions up to 1,400 m^3. The typical use of outdoor systems is to study the ecological effects on plankton communities and in some cases on the marine pelagic system. The duration of the tests ranged from some few days up to several months and is correlated to the size and the complexity of the systems (Figure 4).

Two basic types of marine outdoor microcosms are described in the literature:

- constructed outdoor systems;
- *in situ* enclosures.

Constructed outdoor systems typically consist of moored tanks that contain water, organisms and in some cases a sediment layer. Usually unfiltered sea water is pumped continuously into the microcosms. A good example for such systems are the flow-through experimental tanks of the Marine Ecosystem Research Laboratory of the University of Rhode Island (Vargo *et al*, 1982). The test system consists of a group of 13 m^3 cylinders constructed on the shore of Narragansett Bay, containing water, sediment and a biocoenosis that mimics the biocoenosis of the bay. Waterbody and plankton are continuously exchanged between the cylinders and the sea. To perform the experiments, the cylinders are separated and the test substance continuously added with the flow-through water into the tanks. The reproducibility and the general ecological properties of the systems are described by Oviatt *et al* (1981). The average duration of the tests was about three months.

***In situ* enclosures** are isolated portions of the pelagial. The majority of these systems are floating plastic bags, placed by scuba divers into the water column. Probably the most famous experiments using this design are the floating mesocosms of the Controlled Ecosystem Pollution Experiment (CEPEX) in British Columbia (Menzel and Case, 1977). The used plastic containers contained about 1300 m^3 and extended below the photic zone. They contained three major trophic levels and were used to test the effects of chronic pollution and eutrophication on the marine pelagic biocoenosis. However, due to the large extension, these CEPEX systems are not applicable for the 'routine' ecotoxicological testing of chemicals. Much smaller systems of 1.5 m^3 capacity were described by Kuiper (1977) from the TNO Laboratory for Applied Marine Research in the Netherlands who also published the reproducibility of the experiments. Thereafter a series of different chemicals was tested, and in 1982 he compared the results of laboratory single-species tests with the results in the microcosms (Kuiper, 1982a). He concluded that the optimum dimensions of enclosures for ecotoxicological experiments excluding fish and other large carnivores might be about 2 to 10 m^3 (depending on the oligotrophy of the system). The optimal duration of the tests should not be longer than 4 to 6 weeks, because with time the plankton community diverges more and more from the natural situation.

A total number of 39 chemicals was tested in the different outdoor systems reviewed for this chapter. The major chemical classes tested were:

- general chemicals ~ 35 %
- pesticides ~ 5 %
- heavy metals ~ 30 %

About 30 % of the studies were performed without any perturbator or with variable abiotic factors of ecological relevance.

A great number of other experimental systems are described in the literature. However, only a few of these systems are of appropriate dimension and design for the use in ecotoxicological research. Appropriately designed studies, focusing on the investigation of dose-response relationships for individual and community levels in microcosms, require the reproducibility of the tests and a stable biocoenosis over a suitable testing period. Small systems are usually easier to handle and the composition of the biocoenosis is easier to reproduce. Unfortunately, the degree of similarity with the natural environment and the stability of the community in the microcosms increases with the dimensions of the system. Summarising the literature search (Figure 4), the majority of ecotoxicological studies with benthic communities are performed in systems of < 1 m^3 and with

plankton communities in systems ranging from 1 to 80 m^3. Only a few systems are of appropriate dimensions to study portions of the pelagial. The duration of the tests is variable but generally the greater systems can be used for longer testing periods. Finally, the complexity of the biocoenosis increased with the volume of the different test systems. However, up to a volume of about 10 to 20 m^3 the increase of complexity was significant. Thereafter, the complexity of the systems only increased with the presence of some fish and other large carnivores.

In conclusion, marine microcosms can contribute to characterise the effects of xenobiotics to the marine environment if the studies are focused on key structural or functional endpoints. However, well-designed microcosm studies that can be used to measure these endpoints are extremely cost intensive, and their performance is limited to few laboratories located near a natural marine environment. Therefore, their use in the routine ecotoxicological research might not be appropriate. In special situations, if the risk evaluation of a chemical require such tests, they should clearly be focused on those endpoints that can not be evaluated in conventional laboratory studies.

APPENDIX B. AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

This appendix provides a summary of the available freshwater model ecosystem studies together with the literature references in tabular form (Table B.1). The table is divided into four categories of study design, i.e. in-stream flumes, re-circulating through systems, once-through systems and large outdoor systems.

The order of studies in each category is according to the size of the test system used. The dimensions (L = length, W = width, D = depth) are given in metres; for those studies where details of one or more of the dimensions were not provided the volume (Vol) in litres is given if this was reported. In some study reports only data for length and width were provided.

The column "SS Studies" indicates whether single-species toxicity studies were perfomed for comparison purposes in the frame of the model ecosystem study cited.

TABLE B.1: AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
In-stream flumes								
1.5 x 0.15 x 0.15 m	outdoor	28	21		Medium	trimethyl ammonium chloride, dodecyl	Yes	Lewis *et al*, 1986
1.5 x 0.15 x 0.15 m	outdoor	28	21		Medium	trimethyl ammonium chloride + sewage	Yes	Lewis *et al*, 1986
1.5 x 0.15 x 0.15 m	outdoor	28	21		Medium	linear alkyl benzene sulphonate, dodecyl	Yes	Lewis *et al*, 1993
1.5 x 0.15 x 0.15 m	outdoor	28	21		Medium	linear alkylbenzene sulphonate, dodecyl + effluent	Yes	Lewis *et al*, 1993
Re-circulating through systems								
0.82 x 0.18 x 0.13 m	indoor	14	28		Medium	acid, acid+aluminium		Genter and Amyot, 1994
0.86 x 0.27 x 0.10 m	indoor	0	4		Low	heat		de Koslowski and Bunting, 1981
1.52 x 0.16 x 0.08 m	outdoor	12	20		Medium	zinc sulphate-copper sulphate	Yes	Cherry *et al*, 1988 Farris, 1986
1.52 x 0.16 x 0.08 m	outdoor	7	30		Medium	copper sulphate (2 studies)	Yes	Belanger *et al*, 1990 Cherry *et al*, 1988 Farris *et al*, 1988 Farris *et al*, 1991 Cherry *et al*, 1991
1.52 x 0.16 x 0.10 m	indoor	32	4		Medium	copper sulphate (2 studies)		Clements *et al*, 1988
1.52 x 0.16 x 0.10 m	greenhouse	32	7		Medium	zinc sulphate		Kiffney and Clements, 1994a
1.52 x 0.16 x 0.10 m	greenhouse	94	10		Medium	zinc sulphate + copper sulphate + cadmium sulphate		Kiffney and Clements, 1994b
1.52 x 0.16 x 0.10 m	outdoor	32	10		Medium	copper sulphate		Clements *et al*, 1989
1.52 x 0.16 x 0.15 m	outdoor	13	30		Medium	zinc sulphate (3 studies)	Yes	Belanger *et al*, 1986 Farris *et al*, 1989, 1994 Genter *et al*, 1987, 1988a
1.52 x 0.16 x 0.15 m	outdoor	16	30	34	Medium	zinc sulphate (3 studies)	Yes	Belanger *et al*, 1986 Farris, 1986 Farris *et al*, 1989, 1994

TABLE B.1 (cont.): AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
1.52 x 0.16 x 0.15 m	outdoor	12	30		Medium	zinc sulphate-snail interactions		Genter *et al,* 1988a
1.52 x 0.16 x 0.15 m	outdoor	12	30		Medium	zinc sulphate-pH	Yes	Genter *et al,* 1988b Farris, 1986 Cherry *et al,* 1988
1.70 x 0.24 x 0.13 m	indoor	30	20		Medium	complex industrial effluent	Yes	Pontasch *et al,* 1989a; 1989b Pontasch and Cairns, 1991
1.70 x 0.24 x 0.13 m	indoor	42	30		Medium	fenvalerate		Breneman and Pontasch, 1994
1.70 x 0.24 x 0.14 m	outdoor	32	10		Medium	copper sulphate		Clements *et al,* 1989
2.10 x 0.60 x 0.20 m	indoor	21	28		Low	pH (acid) + aluminium chloride		Burton and Allen, 1986
2.30 x 0.10 m		0	24		Medium	coal leachate		Gerhart *et al,* 1977
2.40 x 0.12 x 0.13 m	outdoor	21	21	28	Medium	atrazine (2 studies)		Kosinksi 1984 Kosinski and Merkle, 1984 Moorhead and Kosinski, 1986
2.40 x 0.12 x 0.13 m	outdoor	21	21		Medium	trifluralin		Kosinski 1984 Kosinski and Merkle, 1984
2.40 x 0.12 x 0.13 m	outdoor	21	21		Medium	MSMA (monosodium methane-arsenate) (2 studies)		Kosinski, 1984 Kosinski and Merkle, 1984
2.40 x 0.12 x 0.13 m	outdoor	21	21		Medium	paraquat		Kosinski, 1984 Kosinski and Merkle, 1984
5 x 0.35 x 0.25 m	outdoor	21	30		High	lindane	Yes	Mitchell *et al,* 1993 Stephenson *et al,* 1992 Crossland, 1988b
5 x 0.35 x 0.25 m	outdoor	21	30		High	petrochemical effluent	Yes	Crossland *et al,* 1992b
5 x 0.35 x 0.25 m	outdoor	21	30		High	linear alkylbenzene sulphonate, dodecyl	Yes	Holt and Mitchell, 1994
6.60 x 0.66 x 0.25 m	greenhouse	162	180		High	sediments		Crouse *et al,* 1981
6.60 x 0.66 x 0.25 m	greenhouse	0	30		High	kraft mill effluent (7 studies)	Yes	Seim *et al,* 1977

TABLE B.1 (cont.): AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
12.20 x 0.30 x 0.40 m	indoor	90	150		High	diflubenzuron	Yes	Hansen and Garton, 1982a; 1982b
22 x 0.30 x 0.05 m					Low	chlorine-grazer interaction		Steinman *et al*, 1992
Once-through systems								
0.33 x 0.13 x 0.11 m	indoor	385	28		Medium	alkyl sulphate, dodecyl		Belanger *et al*, 1996
0.33 x 0.13 x 0.11 m	indoor	350	28		Medium	alkyl ethoxylate sulphate, CL=14.5, EO=2.17		Belanger *et al*, 1996
0.33 x 0.13 x 0.11 m	indoor	45	28		Medium	copper sulphate		Belanger *et al*, 1996
0.50 x 0.39 x 0.05 m	indoor	30	21	14	Medium			Scholz and Müller, 1992
0.35 x 0.28 x 0.15 m	indoor	0	21		High	atrazine		Pratt *et al*, 1988a
0.35 x 0.28 x 0.15 m	indoor	0	10		High	water-soluble fraction/JP-4 jet fuel	Yes	Cairns and Pratt, 1985
0.35 x 0.28 x 0.15 m	indoor	0	28		High	zinc sulphate		Pratt *et al*, 1987a
0.35 x 0.28 x 0.15 m	indoor	21	23		High	zinc (21 d) followed by pH (2 d)		Niederlehner and Cairns, 1993
0.35 x 0.28 x 0.15 m	indoor	0	21		High	copper		Pratt *et al*, 1987b
0.35 x 0.28 x 0.15 m	indoor	0	7		High	Chlorine/NH3/interaction		Cairns *et al*, 1990
0.35 x 0.28 x 0.15 m	indoor	0	28		High	chlorine		Pratt *et al*, 1988b
0.35 x 0.28 x 0.15 m	indoor	0	21		High	selenium		Pratt and Bowers, 1990
0.35 x 0.28 x 0.15 m	indoor	0	21		High	phenol		Pratt *et al*, 1989
0.35 x 0.28 x 0.15 m	indoor	0	21		High	3-trifluoromethyl-4-nitrophenol		McCormick *et al*, 1986
(19 L)	indoor	3	30		Medium	alkylbenzene sulphonate, dodecyl	Yes	Maki, 1980 Larson and Maki, 1982
(19 L)	indoor	3	30		Medium	alkylbenzene sulphonate, dodecyl + sewage	Yes	Maki, 1980 Larson and Maki, 1982
1.20 x 0.10 x 0.01 m	indoor	7	9		Medium	oil shale refinery effluent (6 studies)		Russel *et al*, 1981
1.20 x 0.20 x 0.13 m	outdoor	14	28		Medium	pulp and paper mill effluent	Yes	Amblard *et al*, 1990
1.52 x 0.20 x 0.20 m	outdoor	21	25		High	treated acid mine drainage		Perrin *et al*, 1992 Mundie *et al*, 1991
2 x 0.19 x 0.01 m	outdoor	0	28		Medium	UV-A and B radiation		Bothwell *et al*, 1994

TABLE B.1 (cont.): AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	chlorine as calcium hypochlorite (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	copper sulphate (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	sucrose (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	chromium as potassium dichromate (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	nitrate as ammonium nitrate (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	phosphate as disodium phosphate (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	13	10		Medium	dextrose (2 studies)		Rodgers *et al*, 1980
4 x 0.35 x 0.39 m	outdoor	18	14		Medium	copper (2 studies)		Clark *et al*, 1982
4 x 0.35 x 0.39 m	outdoor	18	14		Medium	chlorine (2 studies)		Clark *et al*, 1982
4 x 0.35 x 0.39 m	outdoor	18	14		Medium	dextrose (2 studies)		Clark *et al*, 1982
4.40 x 0.40 x 0.16 m		0	150		Medium	herbicide contaminated river		Hatakeyama *et al*, 1994
5 x 0.35 x 0.25 m	outdoor	21	30		High	copper	Yes	Stephenson *et al*, 1992
5 x 0.35 x 0.25 m	outdoor	21	30		High	atrazine	Yes	Stephenson *et al*, 1992
5 x 0.35 x 0.25 m	outdoor	21	30		High	3,4-dichloroaniline	Yes	Stephenson *et al*, 1992
6 x 0.20 x 0.20 m	outdoor	40	85		High	pH/Al/interaction (2 studies)		Planas *et al*, 1989 Allard and Moreau, 1987
6.10 x 0.30 x 0.20 m	outdoor	14	98		High	zinc sulphate		Williams and Mount, 1965
6.70 x 0.30 x 0.60 m	outdoor	1095	365		Medium	heat		Wilde and Tilly, 1981
6.80 x 0.60 x 0.10 m pools-1.80 x 1.20 x 0.60 m	outdoor	30	30	15	High	alcohol ethoxylate (A45-EO7) (2 studies)	Yes	Dorn *et al*, 1994a Rodgers *et al*, 1996 Kline *et al*, 1996
6.80 x 0.60 x 0.10 m pools-1.80 x 1.20 x 0.60 m	outdoor	21	30	15	High	alcohol ethoxylate (A91-EO6) (2 studies)	Yes	Dorn *et al*, 1994b Rodgers *et al*, 1996
8 x 0.60 x 0.25 m	indoor	56	5	90	High	3-trifluoromethyl-4-nitrophenol	Yes	Maki *et al*, 1975 Maki and Johnson, 1976, 1977
8 x 0.60 x 0.25 m	indoor	365	30	60	High	atrazine (4 studies)		Lynch *et al*, 1982, 1985
8 x 0.60 x 0.25 m	indoor	90	60		High	2,4,5,2',4',5'-hexachlorobiphenyl (4 studies)		Lynch *et al*, 1982, 1985

TABLE B.1 (cont.): AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
12 x 0.40 x 0.04 m pools 1.20 x 0.40 x 0.10 m	indoor	61	56		High	trimethyl ammonium chloride, dodecyl (2 studies)	Yes	Belanger, 1992, Belanger, 1993a, 1994 Schwab *et al,* 1992 Cuffney *et al,* 1990
12 x 0.40 x 0.04 m pools 1.20 x 0.40 x 0.10 m	indoor	90	56		High	alkyl sulphate, dodecyl	Yes	Belanger and Guckert, 1996 Belanger *et al,* 1995a, 1995b McCormick *et al,* in press Guckert *et al,* 1996
12 x 0.40 x 0.04 m pools 1.20 x 0.40 x 0.10 m	indoor	77	56		High	alkyl ethoxylate sulphate, CL=14.5/EO=2.17	Yes	Belanger *et al,*1995a, 1995b Belanger and Guckert,1996
20 x 0.30 x 0.10 m	indoor	0	120		Medium	trimethyl ammonium chloride, dodecyl	Yes	Woltering and Bishop, 1989
20 x 1.30 x 0.80 m	outdoor	1085	730		High	heat		Bisson and Davis, 1976
22 x 0.30 x 0.05 m	indoor	180	12		Low	chlorine-grazer interaction		Steinman *et al,* 1992
Large outdoor macrocosms								
20 x 1 x 0.25 m	outdoor		49		Medium	sodium chlorate		Matida *et al,* 1975
20 x 1 x 0.25 m	outdoor		49		Medium	sodium sulphamate		Matida *et al,* 1975
45 x 0.40 m	outdoor	171	1	80	High	chlorpyrifos		Pusey *et al,* 1994a, 1994b
50 x 1 x 1 m pools-10 x 2 x 0.30 m (11,250 L)	outdoor	90	42	104	High	triphenyl phosphate contaminated sediments/sediment	Yes	Fairchild *et al,* 1987
50 x 1 x 1 m pools-10 x 2 x 0.30 m	outdoor	30	45		High	linear alkylbenzene sulphonate, dodecyl (11.9)	Yes	Fairchild *et al,* 1993
75 x 0.20 x 0.20 m	outdoor	14	63	56	Medium	cobalt nitrate, copper sulphate, zinc chloride		Eichenberger *et al,* 1981
75 x 0.20 x 0.20 m	outdoor	14	63	56	Medium	copper sulphate		Eichenberger *et al,* 1981
92 x 0.60 x 0.20 m	outdoor	180	333		High	mercuric chloride		Kania *et al,* 1976 Ferens, 1974 Sigmon *et al,* 1977

TABLE B.1 (cont.): AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
92 x 0.60 x 0.20 m	outdoor	150	365	150	High	cadmium chloride	Yes	Bowling *et al,* 1980 Giesy *et al,* 1979, 1981
92 x 0.60 x 0.20 m	outdoor	182	14	15	High	anthracene (3 studies)	Yes	Bowling *et al,* 1983, 1984 Landrum *et al,* 1984 Giesy *et al,* 1983a, 1983b
100 x 0.23 m	outdoor		0.02	21	High	temephos		Yasuno *et al,* 1985
100 x 0.23 m	outdoor		0.02	49	High	chlorphoxim		Yasuno *et al,* 1985
107 x 1.50 x 0.30 m pools-7.90 x 7.90 x 1.20 m	outdoor	365	334		High	bleached kraft mill effluent (7 studies)	Yes	NCASI, 1983
107 x 1.50 x 0.30 m pools-7.90 x 7.90 x 1.20 m	outdoor	365	205	120	High	bleached kraft mill effluent. TCDD/TCDF	Yes	NCASI, 1991
110 x 1.20 x 0.30 m pools- 4 x 3 x 1 m	outdoor	270	510		High	bleached kraft mill effluent (5 studies)	Yes	NCASI, 1985 Hall *et al,* 1991, 1992
110 x 1.20 x 0.30 m pools- 4 x 3 x 1 m	outdoor	270	510		High	bleached kraft mill effluent, ClO2 substitution	Yes	NCASI, 1993
114 x 4.30 x 0.30 m	outdoor	60	134		High	heat		Wrenn *et al,* 1979 Armitage *et al,* 1978, Armitage, 1980 Rodgers, 1980 Wrenn and Granneman, 1980
152 x 1.50 m	outdoor	180	191		High	sucrose (3 studies)		Warren *et al,* 1964
300 x 1.25 x 0.40 m	outdoor		620		High	treated sewage effluent		Watton and Hawkes, 1984
400 x 4 m	outdoor	548	365	270	High	copper sulphate	Yes	Leland *et al,* 1989 Leland and Carter, 1985 Leland and Kent, 1981

TABLE B.1 (cont.): AVAILABLE FLOWING FRESHWATER MODEL ECOSYSTEM STUDIES

Size L x W x D or (Vol)	Location	Pretreatment (d)	Treatment (d)	Recovery (d)	Overall System Complexity	Stressor	SS Studies	Reference
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor	365	730		High	heat	Yes	Arthur *et al,* 1982, Arthur, 1988 Nordlie and Arthur, 1981 Howey and Arthur, 1978
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		119	17	High	acid pH	Yes	McCormick *et al,* 1980, 1989 Zischke *et al,* 1983
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		4	7	High	para-cresol (2 studies)	Yes	Stout and Cooper, 1983 Stout and Kilham, 1983
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		126		High	diazinon	Yes	Arthur *et al,* 1983
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		88	21	High	pentachlorophenol (2 studies)	Yes	Zischke *et al,* 1985 Pignatello *et al,* 1986 Hedtke and Arthur, 1985 Hedtke *et al,* 1986
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		548		High	ammonia	Yes	Arthur *et al,* 1987 Hermanutz *et al,* 1987 Zischke and Arthur, 1987
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		133		High	chlorine	Yes	Newman *et al,* 1987 Hermanutz *et al,* 1987
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		119		High	chlorine/ammonia	Yes	MERS, 1988 Newman and Perry, 1989 Hermanutz *et al,* 1987
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		534		High	selenium as sodium selenite	Yes	Hermanutz *et al,* 1992 Hedtke *et al,* 1989 Pratt and Bowers, 1990 Schultz and Hermanutz, 1990 Allen, 1991
540 x 2 x 0.15 m pools-30 x 3.50 x 0.80 m	outdoor		100		High	chlorpyrifos	Yes	Eaton *et al,* 1985

APPENDIX C. TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
1	FF	outdoor	30	30	15	alcohol ethoxylate (A45-EO7)	68951-67-7	surfactant, nonionic	invertebrate, simuliidae abundance
2	FF	outdoor	30	30	15		68951-67-7	surfactant, nonionic	fish, fathead minnow egg production
3	FF	outdoor	30	30	15		68951-67-7	surfactant, nonionic	fish, fathead minnow survival
4	FF	outdoor	30	30	15		68951-67-7	surfactant, nonionic	fish, fathead minnow larval survival
5	FF	outdoor	21	30	15	alcohol ethoxylate (A91-EO6)	68439-46-3	surfactant, nonionic	fish, fathead minnow egg production
6	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	fish, fathead minnow larval survival
7	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	fish, fathead minnow behaviour
8	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	fish, fathead minnow survival
9	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	invertebrate, *Hyallela azteca* survival
10	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	invertebrate, copepod drift
11	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	invertebrate, cladocera drift
12	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	invertebrate, drift

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0800	0.1600	0.3700		0.7900			fish	0.37	4.63	1)
0.1100	0.2800	0.3700		0.7900			fish	0.37	3.36	1)
0.1600	0.3300	0.3700		0.7900			fish	0.37	2.31	1)
0.2800	0.5500	0.3700		0.7900			fish	0.37	1.32	1)
0.7300	2.0400	1.0100		2.8000			fish	1.01	1.38	2)
0.7300	2.0400	1.0100		2.8000			fish	1.01	1.38	2)
0.7300	2.0400	1.0100		2.8000			fish	1.01	1.38	2)
2.0400	4.3500	1.0100		2.8000			fish	1.01	0.50	2)
2.0400	4.3500	1.0100		2.8000			fish	1.01	0.50	2)
2.0400	4.3500	1.0100		2.8000			fish	1.01	0.50	2)
2.0400	4.3500	1.0100		2.8000			fish	1.01	0.50	2)
2.0400	4.3500	1.0100		2.8000			fish	1.01	0.50	2)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
13	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	invertebrate, *Hyallela azteca* feeding rate
14	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	fish, bluegill survival
15	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	fish, bluegill growth
16	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	algae, periphyton chlorophyll-a
17	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	algae, periphyton dry weight
18	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	algae, periphyton phaeophytin
19	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	algae, periphyton ash-free dry weight
20	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	algae, species composition
21	FF	outdoor	21	30	15		68439-46-3	surfactant, nonionic	algae, periphyton cell leakage
22	FF	indoor	350	28		alkyl ethoxylate sulphate, CL=14.5/EO=2.16(7)	125301-92-0	surfactant, anionic	algae and bacteria, dissolved oxygen evolution
23	FF	indoor	77	56			125301-92-0	surfactant, anionic	algae, density of 2/11 dominant taxa

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
4.3500	5.7000	1.0100		2.8000			fish	1.01	0.23	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
5.7000	11.24	1.0100		2.8000			fish	1.01	0.18	2)
0.0540	0.6080	0.1000		0.2700			fish	0.1	1.85	3)
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
24	FF	indoor	77	56			125301-92-0	surfactant, anionic	invertebrate, corbicula density
25	FF	indoor	77	56			125301-92-0	surfactant, anionic	invertebrate, physa biomass
26	FF	indoor	77	56			125301-92-0	surfactant, anionic	invertebrate, baetis abundance
27	FF	indoor	77	56			125301-92-0	surfactant, anionic	invertebrate, isonychia abundance
28	FF	indoor	77	56			125301-92-0	surfactant, anionic	invertebrate, EPT density
29	FF	indoor	77	56			125301-92-0	surfactant, anionic	invertebrate, EPT biomass
30	FF	indoor	90	56		alkyl sulphate, dodecyl	151-21-3 85711-69-9	surfactant, anionic	algae and bacteria, lipid class partitioning
31	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	algae and bacteria, lipid class profile
32	FF	indoor	385	28			151-21-3 85711-69-9	surfactant, anionic	algae, biovolume of dominant taxa
33	FF	indoor	385	28			151-21-3 85711-69-9	surfactant, anionic	algae, cell density of dominant taxa
34	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	algae, abundance of dominant (2/13) taxa
35	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	algae, density on cobble

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)
0.2510	0.7740	0.1000		0.2700			fish	0.1	0.40	3)
0.0200	0.0610	1.3570	0.4180	3.0000	0.0200		algae	0.02	1.00	4)
0.0200	0.0610	1.3570	0.4180	3.0000	0.0200		algae	0.02	1.00	4)
0.0550	0.1110	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.36	5)
0.0550	0.1110	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.36	5)
0.0610	0.2240	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.33	4)
0.0610	0.2240	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.33	4)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
36	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	algae, biovolume on cobble
37	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	algae, diversity on cobble
38	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, taxa richness
39	FF	indoor	385	28			151-21-3 85711-69-9	surfactant, anionic	algae and bacteria, dissolved oxygen evolution
40	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, oligochaete abundance
41	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, ferrissea abundance
42	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, stenonema abundance
43	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, total abundance
44	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, funcitonal group composition
45	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, corbicula abundance
46	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, physa abundance
47	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	invertebrate, baetis abundance

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0610	0.2240	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.33	4)
0.0610	0.2240	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.33	4)
0.0610	0.2240	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.33	4)
0.1110	0.2740	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.18	5)
0.2240	0.5820	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.09	4)
0.2240	0.5820	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.09	4)
0.2240	0.5820	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.09	4)
0.2240	0.5820	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.09	4)
0.2240	0.5820	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.09	4)
0.5820	1.5860	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.03	4)
0.5820	1.5860	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.03	4)
0.5820	1.5860	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.03	4)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
48	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	bacteria, amino acid uptake
49	FF	indoor	90	56			151-21-3 85711-69-9	surfactant, anionic	bacteria, cell density
50	FF	indoor	28	28	28	alkyl-glucoside, C12/14	110615-47-9	surfactant, anionic	community similarity
51	FF	outdoor		548		ammonia	7664-41-7	inorganic	fish, white sucker growth
52	FF	outdoor		548			7664-41-7	inorganic	invertebrate, sphaerid clam density
53	FF	outdoor		548			7664-41-7	inorganic	fish, walleye survival
54	FF	outdoor		548			7664-41-7	inorganic	fish, white sucker survival
55	FF	outdoor		548			7664-41-7	inorganic	invertebrate, cladoceran density
56	FF	outdoor		548			7664-41-7	inorganic	fish, channel catfsih growth
57	FF	outdoor		548			7664-41-7	inorganic	fish, rainbow trout survival
58	FF	outdoor		548			7664-41-7	inorganic	fish, walleye growth
59	FF	outdoor		548			7664-41-7	inorganic	fish, bluegill growth

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.5820	1.5860	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.03	4)
0.5820	1.5860	1.3570	0.4180	3.0000	0.0200		algae	0.02	0.03	4)
3.6000	5.0000	1.8000		1.0000	2.0000		Daphnia/ inverts	1.0000	0.28	6)
0.0090	0.0400	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.58	7)
0.0100	0.1100	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.52	7)
0.0110	0.1060	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.47	7)
0.0400	0.0900	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.13	7)
0.0480	0.1560	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.11	7)
0.0600	0.1530	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.09	7)
0.0670	0.3290	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.08	7)
0.0990	0.2680	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.05	7)
0.2090	0.4310	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.02	7)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
60	FF	outdoor		548			7664-41-7	inorganic	fish, bluegill survival
61	FF	indoor		21		atrazine	1912-24-9	herbicide	protozoa, dissolved oxygen
62	FF	indoor		21			1912-24-9	herbicide	protozoa, nutrient cycling
63	FF	indoor		21			1912-24-9	herbicide	protozoa, richness
64	FF	indoor		21			1912-24-9	herbicide	protozoa, protein biomass
65	FF	indoor		21			1912-24-9	herbicide	protozoa, chl-a biomass
66	FF	outdoor	21	30			1912-24-9	herbicide	invertebrate, gammarus reproductive behaviour
67	FF	outdoor	21	30			1912-24-9	herbicide	algae, primary productivity
68	FF	outdoor	21	30			1912-24-9	herbicide	algae, community respiration
69	SF	microcosm	42	–	42		1912-24-9	herbicide	primary productivity
70	FF	outdoor	21	30			1912-24-9	herbicide	algae, algal biomass
71	FF	outdoor	21	30			1912-24-9	herbicide	algae, chlorophyll-a

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.2090	0.4310	0.1000	0.0052	0.4200			Daphnia/ inverts	0.0052	0.02	7)
0.0100	0.0320	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.20	8)
0.0100	0.0320	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.20	8)
0.0100	0.0320	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.20	8)
0.0100	0.0320	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.20	8)
0.0100	0.0320	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.20	8)
0.0110	0.0380	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.00	9)
0.0110	0.0380	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.00	9)
0.0110	0.0380	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.00	9)
0.0200	0.1000	0.0650	0.0600	0.1400	0.0220		algae	0.022	1.10	10)
0.0380	0.1200	0.0650	0.0600	0.1400	0.0220		algae	0.022	0.58	9)
0.0380	0.1200	0.0650	0.0600	0.1400	0.0220		algae	0.022	0.58	9)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
72	FF	outdoor	21	21	28		1912-24-9	herbicide	algae abundance
73	FF	outdoor	21	21	28		1912-24-9	herbicide	algae, total abundance
74	SF	microcosm, artificial substrates colonisation test	21	–	28	cadmium	10108-64-2 10124-36-4	metal	protozoan colonisation
75	FF	outdoor	150	365	150		10108-64-2 10124-36-4	metal	algae, diversity
76	FF	outdoor	150	365	150		10108-64-2 10124-36-4	metal	algae, evenness
77	FF	indoor		28		chlorine	7782-50-5	inorganic	protozoa, alkaline phosphatase activity
78	FF	indoor		28			7782-50-5	inorganic	protozoa, species richness
79	FF	indoor		28			7782-50-5	inorganic	protozoa, dissolved oxygen evolution
80	FF	indoor		28			7782-50-5	inorganic	protozoa, protein biomass
81	FF	outdoor		133			7782-50-5	inorganic	macrophyte, chlorosis of leaves
82	FF	outdoor		133			7782-50-5	inorganic	invertebrate, abundance

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.1000	1.0000	0.0650	0.0600	0.1400	0.0220		algae	0.022	0.22	11)
1.0000	10.0	0.0650	0.0600	0.1400	0.0220		algae	0.022	0.02	11)
0.0004	0.0014	0.0310	0.0025	0.0006	0.1000		Daphnia/ inverts	0.0006	1.50	12)
0.0050	0.0100	0.0310	0.0025	0.0006	0.1000		Daphnia/ inverts	0.0006	0.12	13)
0.0050	0.0100	0.0310	0.0025	0.0006	0.1000		Daphnia/ inverts	0.0006	0.12	13)
0.0021	0.0061	0.0110		0.0037			Daphnia/ inverts	0.0037	1.76	14)
0.0021	0.0061	0.0110		0.0037			Daphnia/ inverts	0.0037	1.76	14)
0.0061	0.0250	0.0110		0.0037			Daphnia/ inverts	0.0037	0.61	14)
0.0250	0.1000	0.0110		0.0037			Daphnia/ inverts	0.0037	0.15	14)
0.0520	0.1830	0.0110		0.0037			Daphnia/ inverts	0.0037	0.07	15)
0.0750	0.1830	0.0110		0.0037			Daphnia/ inverts	0.0037	0.05	15)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
83	FF	outdoor		133			7782-50-5	inorganic	bacteria and invertebrate, litter processing
84	FF	indoor		28			7782-50-5	inorganic	protozoa, nutrient sequestering
85	FF	indoor		28			7782-50-5	inorganic	protozoa, ATP biomass
86	FF	indoor		7		chlorine/ ammonia	7782-50-5 7664-41-8	inorganic	protozoa, dissolved oxygen evolution
87	FF	indoor		7			7782-50-5 7664-41-9	inorganic	protozoa, species richness
88	FF	indoor		7			7782-50-5 7664-41-10	inorganic	protozoa, *in vivo* flourescence
89	FF	outdoor		119			7782-50-5 7664-41-7	inorganic	invertebrate, dominant invertebrate biomass
90	FF	outdoor		119			7782-50-5 7664-41-7	inorganic	fish, bluegill mortality
91	FF	outdoor		119			7782-50-5 7664-41-7	inorganic	fish, bluegill growth
92	FF	outdoor		119			7782-50-5 7664-41-7	inorganic	fish, channel catfsih mortality
93	FF	outdoor		119			7782-50-5 7664-41-7	inorganic	invertebrate, dry weight biomass
94	FF	outdoor		119			7782-50-5 7664-41-7	inorganic	bacteria and invertebrate, litter breakdown

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0750	0.1830	0.0110		0.0037			Daphnia/ inverts	0.0037	0.05	15)
0.1000	0.3080	0.0110		0.0037			Daphnia/ inverts	0.0037	0.04	16)
0.1000	0.3080	0.0110		0.0037			Daphnia/ inverts	0.0037	0.04	16)
0.0063	0.0566						–	–	no SS NOEC	17)
0.0063	0.0566						–	–	no SS NOEC	17)
0.0063	0.0566						–	–	no SS NOEC	17)
0.8000	2.4000						–	–	no SS NOEC	18)
0.8000	2.4000						–	–	no SS NOEC	18)
0.8000	2.4000						–	–	no SS NOEC	18)
0.8000	2.4000						–	–	no SS NOEC	18)
0.8000	2.4000						–	–	no SS NOEC	18)
2.4000	2.4000						–	–	no SS NOEC	18)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
95	MA	enclosure	5	–	37	chlorophenol, 4-	106-48-9	organic	plankton, growth and composition
96	FF	outdoor	171	1	80	chlorpyrifos	2921-88-2	insecticide	invertebrate, abundance
97	SF	indoor microcosm	28	–	26/ 15		2921-88-2	insecticide	secondary effects on most other organismic groups
98	FF	outdoor	548	365	270	copper	7758-98-7	metal	invertebrate, drift
99	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, abundance
100	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, secondary production
101	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, total abundance
102	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, taxa richness
103	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, diversity
104	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, similarity
105	FF	outdoor	548	365	270		7758-98-7	metal	invertebrate, annual standing stock
106	SF	indoor microcosm	30	–	224		7758-98-7	metal	DOC and primary production, macroalgal growth

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.1000	1.0000			0.6300	0.32		algae	0.63	3.20	19)
0.0001	0.0026	0.0003		$1.7x10^{-4}$		F+D= 1/10 EC50	Daphnia/ inverts	$1.7x10^{-4}$	1.70	20)
0.0050	0.0350	0.0003		$1.7x10^{-4}$		F+D= 1/10 EC50	Daphnia/ inverts	$1.7x10^{-4}$	0.03	21)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)
0.0040	0.0093	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.33	23)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
107	FF	outdoor	7	30			7758-98-7	metal	algae, bio-accumulation
108	FF	outdoor	7	30			7758-98-7	metal	invertebrate, bioaccumul.
109	FF	outdoor	7	30			7758-98-7	metal	invertebrate, cellulase activity
110	FF	outdoor	7	30			7758-98-7	metal	invertebrate, regulatory capability
111	FF	indoor		21			7758-98-7	metal	protozoa, colonisation rate
112	FF	indoor		21			7758-98-7	metal	protozoa, species richness
113	FF	outdoor	7	30			7758-98-7	metal	invertebrate, mortality
114	FF	outdoor	32	10			7758-98-7	metal	invertebrate, taxon richness
115	FF	indoor	45	28			7758-98-7	metal	algae, total cell density
116	FF	outdoor	21	30			7758-98-7	metal	invertebrate, gammarus drift
117	FF	outdoor	21	30			7758-98-7	metal	invertebrate, gammarus abundance
118	FF	outdoor	21	30			7758-98-7	metal	invertebrate, lymnaea adult biomass

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0055	0.0084	0.0070	0.0013	0.0050	0.0050		Daphnia/ Inverts	0.0013	0.24	24)
0.0055	0.0084	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.24	24)
0.0055	0.0084	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.24	24)
0.0055	0.0084	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.24	24)
0.0066	0.0127	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.20	25)
0.0066	0.0127	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.20	25)
0.0084	0.0139	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.15	24)
0.0090	0.0130	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.14	26)
0.0095	0.0260	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.14	27)
0.0100	0.0250	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.13	28)
0.0100	0.0250	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.13	28)
0.0100	0.0250	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.13	28)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
119	FF	outdoor	21	30			7758-98-7	metal	invertebrate, feeding rate
120	FF	outdoor	21	30			7758-98-7	metal	algae, primary productivity
121	FF	indoor		21			7758-98-7	metal	protozoa, nutrient sequestering
122	FF	indoor	32	4			7758-98-7	metal	invertebrate, taxon diversity
123	FF	indoor		21			7758-98-7	metal	protozoa, autotrophic index
124	FF	outdoor	21	30			7758-98-7	metal	invertebrate, gammarus reproductive behaviour
125	FF	outdoor	21	30			7758-98-7	metal	invertebrate, baetis drift
126	FF	outdoor	21	30			7758-98-7	metal	invertebrate, lymnaea hatching
127	FF	outdoor	21	30			7758-98-7	metal	algae, community composition
128	FF	indoor	45	28			7758-98-7	metal	algae, biovolume of dominant taxa
129	FF	indoor	45	28			7758-98-7	metal	algae, cell density of dominant taxa
130	FF	indoor	45	28			7758-98-7	metal	algae and bacteria, dissolved oxygen evolution

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0100	0.0250	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.13	28)
0.0100	0.0250	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.13	28)
0.0127	0.0195	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.10	25)
0.0150	0.1350	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.09	26)
0.0195	0.0365	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.07	25)
0.0250	0.0860	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	28)
0.0250	0.0860	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	28)
0.0250	0.0860	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	28)
0.0250	0.0860	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	28)
0.0260	0.0470	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	27)
0.0260	0.0470	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	27)
0.0260	0.0470	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	27)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
131	FF	indoor	45	28			7758-98-7	metal	algae, total biovolume density
132	FF	indoor	32	4			7758-98-7	metal	invertebrate, abundance
133	FF	indoor	32	4			7758-98-7	metal	invertebrate, taxon richness
134	FF	indoor	32	4			7758-98-7	metal	invertebrate, total abundance
135	FF	indoor	45	28			7758-98-7	metal	algae and bacteria, ash-free dry weight
136	FF	indoor	45	28			7758-98-7	metal	algae, chlorophyll-a
137	FF	outdoor	21	30			7758-98-7	metal	macrophyte, lemna growth rate
138	FF	indoor	45	28			7758-98-7	metal	algae, community similarity
139	FF	outdoor		126		diazinon	333-41-5	insecticide	invertebrate, total abundance
140	MA	indoor	56	14	–	dibutyl phthalate	84-74-2	organic	individuals, density
141	FF	outdoor	21	30		dichloro-aniline, 3,4-	95-76-1	organic	invertebrate, gammarus reproductive behaviour
142	FF	outdoor	21	30			95-76-1	organic	invertebrate, gammarus abundance

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0260	0.0470	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.05	27)
0.0320	0.1780	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.04	26)
0.0320	0.1780	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.04	26)
0.0320	0.1780	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.04	26)
0.0470	0.0980	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.03	27)
0.0470	0.0980	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.03	27)
0.0860	0.3700	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.02	28)
0.0980	0.2080	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.01	27)
0.0050	0.0120	0.0500	0.0004	0.00064	1.7300	A= 1/10 EC_{50}	Daphnia/ inverts	0.00042	0.08	29)
0.0440	0.4400	0.5600		0.5600	0.2000		algae	0.075	4.55	30)
0.0600	0.1900	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.20	31)
0.2000	0.8000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.06	31)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
143	FF	outdoor	21	30			95-76-1	organic	invertebrate, gammarus drift
144	FF	outdoor	21	30			95-76-1	organic	invertebrate, baetis drift
145	FF	outdoor	21	30			95-76-1	organic	invertebrate, polycelis drift
146	FF	outdoor	21	30			95-76-1	organic	invertebrate, polycelis abundance
147	FF	outdoor	21	30			95-76-1	organic	algae, primary productivity
148	FF	outdoor	21	30			95-76-1	organic	algae, community composition
149	MA	enclosure	5	–	37	dichloro-phenol, 2,4-	120-83-2	organic	plankton, growth and composition
150	FF	indoor	90	150		diflubenz-uron	35367-38-5	insecticide	algae, abundance
151	FF	indoor	90	150			35367-38-5	insecticide	invertebrate, abundance
152	FF	indoor	90	150			35367-38-5	insecticide	invertebrate, biomass
153	FF	indoor	90	150			35367-38-5	insecticide	algae, taxa diversity
154	FF	indoor	90	150			35367-38-5	insecticide	invertebrate, taxa diversity

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.8000	2.4000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.02	31)
0.8000	2.4000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.02	31)
0.8000	2.4000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.02	31)
0.8000	2.4000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.02	31)
0.8000	2.4000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.02	31)
0.8000	2.4000	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.02	31)
0.1000	1.0000			0.3200			Daphnia/ inverts	0.32	3.20	32)
0.0001	0.0100	14.000	0.0001	0.0007		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
0.0001	0.0100	14.000	0.0001	0.0007		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
0.0001	0.0100	14.000	0.0001	0.0007		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
0.0001	0.0100	14.000	0.0001	0.0007		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
0.0001	0.0100	14.000	0.0001	0.0007		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
155	FF	indoor	90	150			35367-38-5	insecticide	bacteria, taxa diversity
156	FF	indoor	90	150			35367-38-5	insecticide	algae, total autotrophic biomass
157	FF	indoor	28	28	28	FAS, C12/14	85586-07-8	surfactant, anionic	community similarity
158	FF	indoor	42	30		fenvalerate	51630-58-1	insecticide	invertebrate, emergence
159	FF	indoor	42	30			51630-58-1	insecticide	invertebrate, taxon richness
160	FF	indoor	42	30			51630-58-1	insecticide	invertebrate, total abundance
161	FF	indoor	42	30			51630-58-1	insecticide	invertebrate, abundance
162	FF	indoor	42	30			51630-58-1	insecticide	invertebrate, drift
163	FF	outdoor	21	30		HCH, gamma-	608-73-1	insecticide	invertebrate, drift
164	FF	outdoor	21	30			608-73-1	insecticide	invertebrate, abundance
165	FF	outdoor	21	30			608-73-1	insecticide	algae, primary productivity
166	FF	indoor	28	28	28	linear alkylbenzene sulphonate, C=11.6	25155-30-0	surfactant, anionic	community similarity

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.001	0.0100	14.000	0.0001	0.0007		F,D=1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
0.0001	0.0100	14.000	0.0001	0.0007		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
1.9400	4.8600	1.7000		16.50	14.400		fish	1.7	0.88	34)
10^{-5}	0.0001	0.0002	$2.3x10^{-5}$			I= 1/10 EC_{50}	Daphnia/ inverts	$2.3x10^{-5}$	2.30	35)
10^{-5}	0.0001	0.0002	$2.3x10^{-5}$			I= 1/10 EC_{50}	Daphnia/ inverts	$2.3x10^{-5}$	2.30	35)
10^{-5}	0.0001	0.0002	$2.3x10^{-5}$			I= 1/10 EC_{50}	Daphnia/ inverts	$2.3x10^{-5}$	2.30	35)
10^{-5}	0.0001	0.0002	$2.3x10^{-5}$			I= 1/10 EC_{50}	Daphnia/ inverts	$2.3x10^{-5}$	2.30	35)
0.0003	0.0029	0.0002	$2.3x10^{-5}$			I= 1/10 EC_{50}	Daphnia/ inverts	$2.3x10^{-5}$	0.08	35)
0.0002	0.0006	0.0088	0.0022	0.0110	1.4000		Daphnia/ inverts	0.0022	11.00	36)
0.0006	0.0031	0.0088	0.0022	0.0110	1.4000		Daphnia/ inverts	0.0022	3.73	36)
0.0006	0.0031	0.0088	0.0022	0.0110	1.4000		Daphnia/ inverts	0.0022	3.73	36)
0.1200	0.3200	0.2300	2.4000	0.3000	0.500		fish	0.23	1.92	37)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
167	FF	indoor	3	30		linear alkylbenzene sulphonate, dodecyl	27176-87-0	surfactant, anionic	fish, bluegill biomass
168	FF	indoor	3	30			27176-87-0	surfactant, anionic	macrophyte, duckweed growth
169	FF	indoor	3	30			27176-87-0	surfactant, anionic	algal community structure
170	FF	indoor	3	30			27176-87-0	surfactant, anionic	invertebrate, daphnia abundance
171	FF	indoor	3	30			27176-87-0	surfactant, anionic	macrophyte, elodea biomass
172	FF	outdoor	21	30			27176-87-0	surfactant, anionic	invertebrate abundance
173	FF	outdoor	21	30			27176-87-0	surfactant, anionic	invertebrate, dominance
174	FF	indoor	3	30			27176-87-0	surfactant, anionic	bacteria, microbial oxygen consumption
175	FF	indoor	3	30			27176-87-0	surfactant, anionic	bacteria, glucose metabolism
176	FF	indoor	3	30			27176-87-0	surfactant, anionic	invertebrate, bluegill biomass
177	FF	outdoor	21	30			27176-87-0	surfactant, anionic	invertebrate, drift
178	FF	indoor	3	30			27176-87-0	surfactant, anionic	bacteria, LAS degradation

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0375	0.0750	0.2300	2.4000	0.3000	0.500		fish	0.23	6.13	38)
0.0375	0.0750	0.2300	2.4000	0.3000	0.500		fish	0.23	6.13	38)
0.0375	0.1500	0.2300	2.4000	0.3000	0.500		fish	0.23	6.13	38)
0.1500	0.3000	0.2300	2.4000	0.3000	0.500		fish	0.23	1.53	38)
0.1500	0.3000	0.2300	2.4000	0.3000	0.500		fish	0.23	1.53	38)
0.3000	1.5000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.77	39)
0.3000	1.5000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.77	39)
0.5000	1.0000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.46	38)
0.5000	1.0000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.46	38)
1.0000	2.0000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.23	38)
1.5000	2.7000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.15	38)
2.0000	4.0000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.12	38)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
179	FF	outdoor	21	30			27176-87-0	surfactant, anionic	invertebrate reproduction
180	FF	outdoor	28	21			27176-87-0	surfactant, anionic	algae, primary productivity
181	FF	outdoor	28	21			27176-87-0	surfactant, anionic	algae, taxa richness
182	FF	outdoor	28	21			27176-87-0	surfactant, anionic	algae, biomass
183	FF	outdoor	28	21			27176-87-0	surfactant, anionic	algae, abundance
184	FF	outdoor	28	21		linear alkylbenzene sulphonate, dodecyl + effluent	27176-87-0	surfactant, anionic	algae, biomass
185	FF	outdoor	28	21			27176-87-0	surfactant, anionic	algae, abundance
186	FF	outdoor	28	21			27176-87-0	surfactant, anionic	algae, primary productivity
187	FF	outdoor	180	333		mercuric chloride	7487-94-7	metal	algae, diversity
188	FF	outdoor	180	333			7487-94-7	metal	invertebrate abundance
189	FF	outdoor	180	333			7487-94-7	metal	algae, algal colonisation
190	FF	outdoor	180	333			7487-94-7	metal	invertebrate diversity
191	SF	experi-mental ponds	60	14	120	metamitron	41394-05-2	herbicide	phytoplankton & Cladocera, population dynamics
192	SF	pond meso-cosm	150		180	metolachlor	51212-45-2	herbicide	primary production, effects on populations

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
2.7000	3.0000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.09	39)
1.1000	9.8000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.21	40)
1.1000	9.6000	0.2300	2.4000	0.3000	0.500		fish	0.23	0.21	40)
9.8000	28.10	0.2300	2.4000	0.3000	0.500		fish	0.23	0.02	40)
9.8000	28.10	0.2300	2.4000	0.3000	0.500		fish	0.23	0.02	40)
9.8000	28.10	0.2300	2.4000	0.3000	0.500		fish	0.23	0.02	40)
9.8000	28.10	0.2300	2.4000	0.3000	0.500		fish	0.23	0.02	40)
9.8000	28.10	0.2300	2.4000	0.3000	0.500		fish	0.23	0.02	40)
0.0001	0.0010	0.0009		0.0050	0.0080	A=1/10 EC_{50}	fish	0.0009	9.00	41)
0.0010	0.0010	0.0009		0.0050	0.0080	A=1/10 EC_{50}	fish	0.0009	0.90	41)
0.0010	0.0050	0.0009		0.0050	0.0080	A=1/10 EC_{50}	fish	0.0009	0.90	41)
0.0010	0.0050	0.0009		0.0050	0.0080	A=1/10 EC_{50}	fish	0.0009	0.90	41)
0.4000	2.0000	32.60		32.00	0.3000	F=1/10 LC_{50}	algae	0.3	0.75	42)
0.5000	1.0000	0.7800		5.9000	0.0100		algae	0.01	0.02	43)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
193	SF	in-situ limno-corrals outdoor	30	–	119	methoxy-chlor	72-43-5	insecticide	species composition & community structure of aquatic organisms
194	FF	outdoor	21	21		MSMA	2163-80-6	herbicide	algae abundance
195	FF	outdoor	21	21			2163-80-6	herbicide	algae, total biovolume
196	FF	outdoor	21	21		paraquat	1910-42-5	herbicide	algae, abundance
197	FF	outdoor		88	21	pentachloro-phenol	87-86-5	fungicide	fish, fathead minnow survival
198	FF	outdoor		88	21		87-86-5	fungicide	fish, fathead minnow growth
199	FF	outdoor		88	21		87-86-5	fungicide	fish, fathead minnow reproduction
200	FF	outdoor		88	21		87-86-5	fungicide	invertebrate, abundance-dominant cladocera
201	FF	outdoor		88	21		87-86-5	fungicide	algae and bacteria, diel metabolism-DO
202	FF	outdoor		88	21		87-86-5	fungicide	algae, periphyton ATP biomass
203	FF	outdoor		88	21		87-86-5	fungicide	algae, chlorophyll-a periphyton
204	FF	outdoor		88	21		87-86-5	fungicide	fish, fathead minnow larval drift
205	FF	outdoor		88	21		87-86-5	fungicide	invertebrate, physa abundance

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0030	0.0050	0.0052	0.0004	0.0008		F,D= 1/10 EC_{50}	Daphnia/ inverts	0.00078	0.13	44)
0.10	1.00			7.7500		D=1/10 EC_{50}	Daphnia/ inverts	7.75	77.50	45)
1.00	10.00			7.7500		D=1/10 EC_{50}	Daphnia/ inverts	7.75	7.75	45)
0.100	1.0000				0.0050		algae	0.0050	0.05	46)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.10	47)
0.1440	0.4320	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.03	47)
0.1440	0.4320	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.03	47)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
206	FF	outdoor		88	21		87-86-5	fungicide	invertebrate, physa reproduction
207	FF	outdoor		88	21		87-86-5	fungicide	invertebrate, diversity
208	FF	indoor		21		phenol	108-95-2	organic	protozoa, dissolved oxygen evolution
209	FF	indoor		21			108-95-2	organic	protozoa, chlorophyll biomass
210	FF	indoor		21			108-95-2	organic	protozoa, dissolved oxygen evolution
211	FF	indoor		21			108-95-2	organic	protozoa, chlorophyll biomass
212	FF	indoor		21			108-95-2	organic	protozoa, species richness
213	FF	indoor		21		selenium	7782-49-2	metal	protozoa, hexosamine biomass
214	FF	indoor		21			7782-49-2	metal	protozoa, production biomass
215	FF	indoor		21			7782-49-2	metal	protozoa, chlorophyll-a
216	FF	indoor		21			7782-49-2	metal	protozoa, species richness
217	SF	pond meso-cosm	275		273	terbuthyl-azine	5914-41-3	herbicide	primary production
218	SF	pond meso-cosm	275		273		5914-41-3	herbicide	nutrients concentration

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.1440	0.4320	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.03	47)
0.1440	0.4320	0.0047	0.0550	0.0750	0.0080	A=1/10 EC_{50}	fish	0.0047	0.03	47)
1.2000	7.7000	0.7500			5.0000		fish	0.75	0.63	48)
1.2000	7.7000	0.7500			5.0000		fish	0.75	0.63	48)
3.0000	11.000	0.7500			5.0000		fish	0.75	0.25	48)
3.0000	11.000	0.7500			5.0000		fish	0.75	0.25	48)
3.0000	11.000	0.7500			5.0000		fish	0.75	0.25	48)
0.0201	0.0411	0.0280	0.1300	0.3000			fish	0.028	1.39	49)
0.0411	0.0849	0.0280	0.1300	0.3000			fish	0.028	0.68	49)
0.0411	0.0849	0.0280	0.1300	0.3000			fish	0.028	0.68	49)
0.0411	0.0849	0.0280	0.1300	0.3000			fish	0.028	0.68	49)
0.0050	0.0100	0.2380		0.2100	0.0033		algae	0.0033	0.66	50)
0.0100	0.0200	0.2380		0.2100	0.0033		algae	0.0033	0.33	50)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
219	SF	pond meso-cosm	275		273		5914-41-3	herbicide	chlorophy-caea, No. of species
220	SF	pond meso-cosm	275		273		5914-41-3	herbicide	macrophytes
221	SF	pond meso-cosm	275		273		5914-41-3	herbicide	phytoplankto n diversity
222	FF	outdoor	28	21		trimethyl ammonium chloride, dodecyl + sewage	112-00-5	surfactant, cationic	algae, biomass
223	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, primary productivity
224	FF	indoor	61	56		trimethyl ammonium chloride, dodecyl	112-00-5	surfactant, cationic	invertebrate, ferrissea abundance
225	FF	indoor	61	56			112-00-5	surfactant, cationic	algae, total abundance
226	FF	indoor	61	56			112-00-5	surfactant, cationic	algae, community similarity
227	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, primary productivity
228	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, community similarity
229	FF	indoor	61	56			112-00-5	surfactant, cationic	invertebrate, corbicula biomass
230	FF	indoor	61	56			112-00-5	surfactant, cationic	invertebrate, mayfly abundance
231	FF	indoor	61	56			112-00-5	surfactant, cationic	algae, primary productivity

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0200	0.0500	0.2380		0.2100	0.0033		algae	0.0033	0.17	50)
0.0500	0.1000	0.2380		0.2100	0.0033		algae	0.0033	0.07	50)
0.0500	0.1000	0.2380		0.2100	0.0033		algae	0.0033	0.07	50)
0.9600	6.9000	0.4600	0.1200	0.0650			Daphnia/ inverts	0.065	0.07	51)
0.9600	6.9000	0.4600	0.1200				Daphnia/ inverts	0.12	0.13	51)
0.0500	0.2350	0.4600	0.1200				Daphnia/ inverts	0.12	2.40	52)
0.0500	0.2350	0.4600	0.1200				Daphnia/ inverts	0.12	2.40	52)
0.0500	0.1850	0.4600	0.1200				Daphnia/ inverts	0.12	2.40	52)
0.2100	0.9600	0.4600	0.1200				Daphnia/ inverts	0.12	0.57	51)
0.2100	0.9600	0.4600	0.1200				Daphnia/ inverts	0.12	0.57	51)
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	51)
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	52)
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	52)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
232	FF	indoor	61	56			112-00-5	surfactant, cationic	algae, taxa richness
233	FF	indoor	61	56			112-00-5	surfactant, cationic	algae, taxa diversity
234	FF	indoor	61	56			112-00-5	surfactant, cationic	invertebrate, EPT abundance
235	FF	indoor	61	56			112-00-5	surfactant, cationic	algae, chlorophyll-a
236	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, biomass
237	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, abundance
238	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, taxa richness
239	FF	outdoor	28	21			112-00-5	surfactant, cationic	algae, community density
240	FF	indoor		120			112-00-5	surfactant, cationic	invertebrate, daphnia density
241	FF	indoor		28		zinc	7733-02-0 7646-85-7	metal	protozoa *in vivo* flourescence
242	FF	indoor		28			7733-02-0 7646-85-7	metal	protozoa, species richness
243	FF	outdoor	13	30			7733-02-0 7646-85-7	metal	invertebrate, bioaccumulat.
244	FF	outdoor	13	30			7733-02-0 7646-85-7	metal	invertebrate, cellulase activity
245	FF	indoor		28			7733-02-0 7646-85-7	metal	protozoa, alkaline phosphatase activity

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	52)
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	52)
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	52)
0.2350	1.1510	0.4600	0.1200				Daphnia/ inverts	0.12	0.51	52)
0.9600	6.9000	0.4600	0.1200				Daphnia/ inverts	0.12	0.13	51)
6.9000	11.60	0.4600	0.1200				Daphnia/ inverts	0.12	0.02	51)
6.9000	11.60	0.4600	0.1200				Daphnia/ inverts	0.12	0.02	51)
6.9000	11.60	0.4600	0.1200				Daphnia/ inverts	0.12	0.02	51)
0.1100	0.1800	0.4600	0.1200				Daphnia/ inverts	0.12	1.09	53)
0.0042	0.0107	0.0960	0.1200				fish	0.096	22.86	54)
0.0107	0.0298	0.0960	0.1200				fish	0.096	8.97	54)
0.0200	0.0340	0.0960	0.1200				fish	0.096	4.80	55)
0.0200	0.0340	0.0960	0.1200				fish	0.096	4.80	55)
0.0298	0.0892	0.0960	0.1200				fish	0.096	3.22	54)

APPENDIX C (cont.). TOTAL DATABASE EVALUATED IN SINGLE-SPECIES

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
246	FF	outdoor	13	30			7733-02-0 7646-85-7	metal	invertebrate, growth
247	FF	outdoor	13	30			7733-02-0 7646-85-7	metal	algae, community biovolume
248	FF	outdoor	13	30			7733-02-0 7646-85-7	metal	invertebrate, mortality

(SS) VERSUS MULTI-SPECIES (MS) COMPARISONS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0340	0.5040	0.0960	0.1200				fish	0.096	2.82	55)
0.0540	0.5040	0.0960	0.1200				fish	0.096	1.78	55)
0.5040	0.9840	0.0960	0.1200				fish	0.096	0.19	55)

REFERENCES

(M = Multispecies Test; F = Fish, I = Invertebrates, D = Daphnia, A = algae, ch = chronic)

1) M = Dorn *et al,* 1996a and 1996b; Kline *et al,* 1996; Gillespie *et al,* 1996a; F = Lizotte *et al,* 1996; D = Gillespie *et al,* 1996b. 2) M = Dorn *et al,* 1994 and 1996b; F = Lizotte *et al,* 1996; D = Gillespie *et al,* 1996b. 3) M = Belanger *et al,* 1996; F,D(ch) = Maki 1979. 4) M,F,D,I (ch)= Belanger and Guckert 1996; Belanger *et al,* 1995b. 5) M,F,D,I(ch) = Belanger *et al,* 1996. 6) M,F,D,A = Steber *et al,* 1995. 7) M = Arthur *et al,* 1987; Hermanutz *et al,* 1987; Zischke *et al,* 1987; F,I,D = ECETOC, 1993b. 8) M = Pratt *et al,* 1988a; F,I,D = ECETOC, 1993b; A = Schäfer *et al,* 1994. 9) M,A = Stephenson *et al,* 1992; F,I,D = ECETOC, 1993b. 10) M = Stay *et al,* 1989; F,I,D = ECETOC, 1993b; A = Stephenson *et al,* 1992. 11) M = Kosinksi, 1984; Kosinski and Merkle, 1984; Moorhead and Kosinski, 1986; F,I,D=ECETOC, 1993b; A = Stephenson *et al,* 1992. 12) M = Cairns and Pratt, 1985; F,I,D = ECETOC, 1993b; A = Kühn and Pattard, 1990. 13) M = Bowling *et al,* 1980; Giesy *et al,* 1979; 1981; F,I,D = ECETOC, 1993b; A = Kühn and Pattard 1990. 14) M = Pratt *et al,* 1988b; F,D = Arthur, 1975. 15) M = Newman *et al,* 1987; Hermanutz *et al,* 1988; F,D = Arthur, 1975. 16) M = Pratt *et al,* 1988b; F,D = Arthur, 1975; A = Murray 1979. 17) Cairns *et al,* 1990. 18) MERS, 1988; Newman *et al,* 1989; Hermanutz *et al,* 1988. 19) M = Kuiper and Hanstveit, 1984a; D = ECETOC, 1993b; A = Kuiper 1982. 20) M = Pusey *et al,* 1994a; 1994b; F,D = Tomlin, 1994. 21) M = Brock *et al,* 1992; F,D = Tomlin, 1994. 22) M = Leland *et al,* 1989; Leland and Carter 1985; Leland and Kent 1985; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 23) M = Hedtke 1984; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 24) M = Belanger *et al,* 1990; Cherry *et al,* 1988; Farris *et al,* 1988; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 25) M = Pratt *et al,* 1987b; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 26) M = Clements *et al,* 1989a; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 27) M=Belanger *et al,* 1996; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 28) M,D,A = Stephenson *et al,* 1992; F,I = ECETOC, 1993b. 29) M = Arthur *et al,* 1983; F = ECETOC, 1993b; A = Hitz, 1981; D = Vial, 1990; I=Hidetsuga, 1972. 30) M=Tagatz *et al,* 1983; F,D = ECETOC, 1993b; A = Acey *et al,* 1987. 31) M,A = Stephenson *et al,* 1992; D = ECETOC, 1993b; F = Nagel *et al,* 1991. 32) M = Kuiper and Hanstveit, 1984a; D = Kühn *et al,* 1989. 33) M = Hansen and Garton, 1982a; 1982b; F,D = Tomlin, 1994; I = Miura and Takahashi, 1974. 34) M,F,D,A = Guhl, 1996b. 35) M = Breneman and Pontasch, 1994; F = ECETOC, 1993b; I = Tomlin 1994. 36) M = Mitchell *et al,* 1993; F,D,I = ECETOC, 1993b; A = Stephenson *et al,* 1992. 37) M = Tattersfield *et al,* 1995; Van de Plassche *et al,* 1997; F,I,D,A = Feijtel and van der Plassche, 1995; D,A = Guhl and Gode, 1989. 38) M,I = Maki, 1980; Larson and Maki, 1982; F,I,D,A = Feijtel and van der Plassche, 1995; D,A = Guhl and Gode, 1989. 39) M = Holt and Mitchell, 1994; F,I,D,A = Feijtel and van der Plassche, 1995; D,A = Guhl and Gode, 1989. 40) M = Lewis *et al,* 1993; F,I,D,A = Feijtel and van der Plassche, 1995; D,A = Guhl and Gode, 1989. 41) M,A = Kania *et al,* 1976; Ferens, 1974; Sigmon *et al,* 1977; F = Call and Geiger, 1992; D = Adams and Heidolph, 1985. 42) M,A,F,D = Heimbach *et al,* 1994. 43) M = Huber, 1995a; F = Dionne, 1978; D = Putt, 1995; A = Hoberg, 1995. 44) M = Stephenson *et al,* 1986; F,D = Tomlin, 1994; I = ECETOC, 1993b. 45) M,D = Kosinksi, 1984; Kosinksi and Merkle, 1984. 46) M = Kosinksi, 1984; Kosinksi and Merkle, 1984; A= Walsh 1972. 47) M = Zischke *et al,* 1985; Pignatello *et al,* 1986; Hedtke and Arthur, 1985; Hedtke *et al,* 1986; F,I,D = ECETOC, 1993b; A = Adema and Vink, 1981. 48) M = Pratt *et al,* 1989; F = ECETOC, 1993b; A = Lauth *et al,* 1990. 49) M = Pratt and Bowers, 1990; F = Adams, 1976; D = Ogle and Wright, 1996; I= Brasher and Ogle, 1993. 50) M = Huber, 1995b; F = Ritter, 1990; D = Rufli, 1989; A = Grade, 1993. 51) M = Lewis *et al,* 1986; F,I = Belanger, 1992; Belanger *et al,* 1993, 1994. 52) M = Belanger, 1992; Belanger *et al,* 1993, 1994; F,I = Belanger, 1992. 53) M = Woltering and Bishop, 1989; F,I = Belanger, 1992; Belanger *et al,* 1993, 1994. 54) M = Pratt *et al,* 1987a; F,I = ECETOC, 1993b. 55) M = Belanger *et al,* 1986; Farris *et al,* 1989, 1994; Genter *et al,* 1987, 1988a; F,I = ECETOC, 1993b.

APPENDIX D. ENTRIES OF THE DATABASE (APPENDIX C) USED

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
1	FF	outdoor	30	30	15	alcohol ethoxylate (A45-EO7)	68951-67-7	surfactant, nonionic	invertebrate, simuliidae abundance
5	FF	outdoor	21	30	15	alcohol ethoxylate (A91-EO6)	68439-46-3	surfactant, nonionic	fish, fathead minnow egg production
22	FF	indoor	350	28		alkyl ethoxylate sulphate, CL=14.5/EO =2.16(7)	125301-92-0	surfactant, anionic	algae and bacteria, dissolved oxygen evolution
30	FF	indoor	90	56		alkyl sulphate, dodecyl	151-21-3 85711-69-9	surfactant, anionic	algae and bacteria, lipid class partitioning
50	FF	indoor	28	28	28	alkyl-glucoside, C12/14	110615-47-9	surfactant, anionic	community similarity
51	FF	outdoor		548		ammonia	7664-41-7	inorganic	fish, white sucker growth
61	FF	indoor		21		atrazine	1912-24-9	herbicide	protozoa, dissolved oxygen
74	SF	microcosm, artificial substrates colonisation test	21	–	28	cadmium	10108-64-2 10124-36-4	metal	protozoan colonisation
77	FF	indoor		28		chlorine	7782-50-5	inorganic	protozoa, alkaline phosphatase activity
95	MA	enclosure	5	–	37	chlorophenol 4-	106-48-9	organic	plankton, growth and composition
96	FF	outdoor	171	1	80	chlorpyrifos	2921-88-2	insecticide	invertebrate, abundance
98	FF	outdoor	548	365	270	copper	7758-98-7	metal	invertebrate, drift

FOR CALCULATION OF FACTORS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0800	0.1600	0.3700		0.7900			fish	0.37	4.63	1)
0.7300	2.0400	1.0100		2.8000			fish	1.01	1.38	2)
0.0540	0.6080	0.1000		0.2700			fish	0.1	1.85	3)
0.0200	0.0610	1.3570	0.4180	3.0000	0.0200		algae	0.02	1.00	4)
3.6000	5.0000	1.8000		1.0000	2.0000		Daphnia/ inverts	1.0000	0.28	6)
0.0090	0.0400	0.1000	0.0052	0.4200			Daphnia/ inverters	0.0052	0.58	7)
0.0100	0.0320	0.0650	0.0600	0.1400	0.0220		algae	0.022	2.20	8)
0.0004	0.0014	0.0310	0.0025	0.0006	0.1000		Daphnia/ inverts	0.0006	1.50	12)
0.0021	0.0061	0.0110		0.0037			Daphnia/ inverts	0.0037	1.76	14)
0.1000	1.0000			0.6300	0.32		Algae	0.63	3.20	19)
0.0001	0.0026	0.0003		1.7×10^{-4}		F + D = 1/10 EC_{50}	Daphnia/ inverts	1.7×10^{-4}	1.70	20)
0.0025	0.0070	0.0070	0.0013	0.0050	0.0050		Daphnia/ inverts	0.0013	0.52	22)

APPENDIX D (cont.). ENTRIES OF THE DATABASE (APPENDIX C) USED

No.	Test type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
139	FF	outdoor		126		diazinon	333-41-5	insecticide	invertebrate, total abundance
140	MA	indoor	56	14	–	dibutyl phthalate	84-74-2	organic	individuals, density
141	FF	outdoor	21	30		dichloro-aniline, 3,4-	95-76-1	organic	invertebrate, gammarus reproductive behaviour
149	MA	enclosure	5	–	37	dichloro-phenol, 2,4-	120-83-2	organic	plankton, growth and composition
150	FF	indoor	90	150		diflubenz-uron	35367-38-5	insecticide	algae, abundance
157	FF	indoor	28	28	28	FAS, C12/14	85586-07-8	surfactant, anionic	community similarity
158	FF	indoor	42	30		fenvalerate	51630-58-1	insecticide	invertebrate, emergence
163	FF	outdoor	21	30		HCH, gamma-	608-73-1	insecticide	invertebrate, drift
166	FF	indoor	28	28	28	linear alkyl benzene sulphonate c=11.6	25155-30-0	surfactant, anionic	community similarity
167	FF	indoor	3	30		linear alkyl benzene sulphonate dodecyl	27176-87-0	surfactant, anionic	fish, bluegill biomass
187	FF	outdoor	180	333		mercuric chloride	7487-94-7	metal	algae, diversity

FOR CALCULATION OF FACTORS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.0050	0.0120	0.0500	0.0004	0.00064	1.7300	A = 1/10 EC_{50}	Daphnia/ inverts	0.00042	0.08	29)
0.0440	0.4400	0.5600		0.5600	0.2000		algae	0.075	4.55	30)
0.0600	0.1900	0.0200		0.0120	0.5000		Daphnia/ inverts	0.012	0.20	31)
0.1000	1.0000			0.3200			Daphnia/ inverts	0.32	3.20	32)
0.0001	0.0100	14.000	0.0001	0.0007		F, D = 1/10 EC_{50}	Daphnia/ inverts	0.0001	1.00	33)
1.9400	4.8600	1.7000		16.50	14.400		fish	1.7	0.88	34)
10^{-5}	0.0001	0.0002	$2.3x10^{-5}$			I = 1/10 EC_{50}	Daphnia/ inverts	$2.3x10^{-5}$	2.30	35)
0.0002	0.0006	0.0088	0.0022	0.0110	1.4000		Daphnia/ inverts	0.0022	11.00	36)
0.1200	0.3200	0.2300	2.4000	0.3000	0.500		fish	0.23	1.92	37)
0.0375	0.0750	0.2300	2.4000	0.3000	0.500		fish	0.23	6.13	38)
0.0001	0.0010	0.0009		0.0050	0.0080	A = 1/10 EC_{50}	fish	0.0009	9.00	41)

APPENDIX D (cont.). ENTRIES OF THE DATABASE (APPENDIX C) USED

No.	Test Type	Location	Duration [d]			Substance			MS Test, Endpoint
			pre-treatment	treatment	post-treatment	Name	CAS-No.	ECETOC Classification	Description
191	SF	experimental ponds	60	14	120	metamitron	41394-05-2	herbicide	phytoplankton & Cladocera, population dynamics
192	SF	pond meso-cosm	150		180	metolachlor	51212-45-2	herbicide	primary production, effects on populations
193	SF	in-situ limno-corrals outdoor	30	–	119	methoxy-chlor	72-43-5	insecticide	species composition & community structure of aquatic organisms
194	FF	outdoor	21	21		MSMA	2163-80-6	herbicide	algae abundance
196	FF	outdoor	21	21		paraquat	1910-42-5	herbicide	algae, abundance
197	FF	outdoor		88	21	pentachloro-phenol	87-86-5	fungicide	fish, fathead minnow survival
208	FF	indoor		21		phenol	108-95-2	organic	protozoa, dissolved oxygen evolution
213	FF	indoor		21		selenium	7782-49-2	metal	protozoa, hexosamine biomass
217	SF	pond meso-cosm	275		273	terbuthyl-azine	5914-41-3	herbicide	primary production
224	FF	indoor	61	56		trimethyl ammonium chloride, dodecyl	112-00-5	surfactant, cationic	invertebrate, ferrissea abundance
241	FF	indoor		28		zinc	7733-02-0 7646-85-7	metal	protozoa *in vivo* flourescence

FOR CALCULATION OF FACTORS

MS Test, Effect Concentration		SS Values					Evaluation			Ref.
NOEC [mg/l]	LOEC [mg/l]	Fish NOEC [mg/l]	Inverts NOEC [mg/l]	Daph. NOEC [mg/l]	Algae NOEC [mg/l]	Remarks	Most sens. SS	Most sens. SS NOEC	Most sens. SS NOEC/ MS NOEC	
0.4000	2.0000	32.60		32.00	0.3000	F = 1/10 LC_{50}	algae	0.3	0.75	42)
0.5000	1.0000	0.7800		5.9000	0.0100		algae	0.01	0.02	43)
0.0030	0.0050	0.0052	0.0004	0.0008		F, D = 1/10 EC_{50}	Daphnia/ inverts	0.00078	0.13	44)
0.10	1.00			7.7500		D = 1/10 EC_{50}	Daphnia/ inverts	7.75	77.50	45)
0.100	1.0000				0.0050		algae	0.0050	0.05	46)
0.0480	0.1440	0.0047	0.0550	0.0750	0.0080	A = 1/10 EC_{50}	fish	0.0047	0.10	47)
1.2000	7.7000	0.7500			5.0000		fish	0.75	0.63	48)
0.0201	0.0411	0.0280	0.1300	0.3000			fish	0.028	1.39	49)
0.0050	0.0100	0.2380		0.2100	0.0033		algae	0.0033	0.66	50)
0.0500	0.2350	0.4600	0.1200				Daphnia / inverts	0.12	2.40	52)
0.0042	0.0107	0.0960	0.1200				fish	0.096	22.86	54)

REFERENCES

(M = Multispecies Test; F = Fish, I = Invertebrates, D = Daphnia, A = algae, ch = chronic)

1) M = Dorn *et al,* 1996a and 1996b; Kline *et al,* 1996; Gillespie *et al,* 1996a; F = Lizotte *et al,* 1996; D = Gillespie *et al,* 1996b. 2) M = Dorn *et al,* 1994 and 1996b; F = Lizotte *et al,* 1996; D = Gillespie *et al,* 1996b. 3) M = Belanger *et al,* 1996; F,D(ch) = Maki 1979. 4) M,F,D,I (ch)= Belanger and Guckert 1996; Belanger *et al,* 1995b. 6) M,F,D,A = Steber *et al,* 1995. 7) M = Arthur *et al,* 1987; Hermanutz *et al,* 1987; Zischke *et al,* 1987; F,I,D = ECETOC, 1993b. 8) M = Pratt *et al,* 1988a; F,I,D = ECETOC, 1993b; A = Schäfer *et al,* 1994. 12) M = Cairns and Pratt, 1985; F,I,D = ECETOC, 1993b; A = Kühn and Pattard, 1990. 14) M = Pratt *et al,* 1988b; F,D=Arthur, 1975. 19) M = Kuiper and Hanstveit, 1984a; D = ECETOC, 1993b; A = Kuiper 1982. 20) M = Pusey *et al,* 1994a; 1994b; F,D = Tomlin, 1994. 22) M = Leland *et al,* 1989; Leland and Carter 1985; Leland and Kent 1985; F,I = ECETOC, 1993b; D,A = Stephenson *et al,* 1992. 28) M,D,A = Stephenson *et al,* 1992; F,I = ECETOC, 1993b. 29) M = Arthur *et al,* 1983; F = ECETOC, 1993b; A = Hitz, 1981; D = Vial, 1990; I=Hidetsuga, 1972. 30) M=Tagatz *et al,* 1983; F,D = ECETOC, 1993b; A = Acey *et al,* 1987. 31) M,A = Stephenson *et al,* 1992; D = ECETOC, 1993b; F = Nagel *et al,* 1991. 32) M = Kuiper and Hanstveit, 1984a; D = Kühn *et al,* 1989. 33) M = Hansen and Garton, 1982a; 1982b; F,D = Tomlin, 1994; I = Miura and Takahashi, 1974. 34) M,F,D,A = Guhl, 1996b. 35) M = Breneman and Pontasch, 1994; F = ECETOC, 1993b; I = Tomlin 1994. 36) M = Mitchell *et al,* 1993; F,D,I = ECETOC, 1993b; A = Stephenson *et al,* 1992. 37) M = Tattersfield *et al,* 1995; Van de Plassche *et al,* 1997; F,I,D,A = Feijtel and van der Plassche, 1995; D,A = Guhl and Gode, 1989. 38) M,I = Maki, 1980; Larson and Maki, 1982; F,I,D,A = Feijtel and van der Plassche, 1995; D,A = Guhl and Gode, 1989. 41) M,A = Kania *et al,* 1976; Ferens, 1974; Sigmon *et al,* 1977; F = Call and Geiger, 1992; D = Adams and Heidolph, 1985. 42) M,A,F,D = Heimbach *et al,* 1994. 43) M = Huber, 1995a; F = Dionne, 1978; D = Putt, 1995; A = Hoberg, 1995. 44) M = Stephenson *et al,* 1986; F,D = Tomlin, 1994; I = ECETOC, 1993b. 45) M,D = Kosinksi, 1984; Kosinksi and Merkle, 1984. 46) M = Kosinksi, 1984; Kosinksi and Merkle, 1984; A= Walsh 1972. 47) M = Zischke *et al,* 1985; Pignatello *et al,* 1986; Hedtke and Arthur, 1985; Hedtke *et al,* 1986; F,I,D = ECETOC, 1993b; A = Adema and Vink, 1981. 48) M = Pratt *et al,* 1989; F = ECETOC, 1993b; A = Lauth *et al,* 1990. 49) M = Pratt and Bowers, 1990; F = Adams, 1976; D = Ogle and Wright, 1996; I= Brasher and Ogle, 1993. 50) M = Huber, 1995b; F = Ritter, 1990; D = Rufli, 1989; A = Grade, 1993. 52) M = Belanger, 1992; Belanger *et al,* 1993, 1994; F,I = Belanger, 1992. 54) M = Pratt *et al,* 1987a; F,I = ECETOC, 1993b. 55) M = Belanger *et al,* 1986; Farris *et al,* 1989, 1994; Genter *et al,* 1987, 1988a; F,I = ECETOC, 1993b.

BIBLIOGRAPHY

Abbott W, 1966. Microcosm studies on estuarine waters. 1. The replicability of microcosms. J. Water Pollut. Control Fed. 38, 258-270.

Acey R, Healy P, Unger TF, Ford CE and Hudson RA, 1987. Growth and aggregation behavior of representative phytoplankton as affected by the environmental contaminant di-n-butyl phthalate. Bull. Environ. Contam. Toxicol. 39, 1-6.

Adams SM and Giddings JM, 1982. Review and evaluation of microcosms for assessing effects of stress in marine ecosystems. Environment International 7, 409-418.

Adams WJ, 1976. The toxicity and residue dynamics of selenium in fish and aquatic invertebrates. PhD Thesis, Michigan State University, East Lansing, Michigan.

Adams WJ and Heidolph BB, 1985. Short-cut chronic toxicity estimates using *Daphnia magna*. Seventh Symposium, ASTM STP 854, Cardwell RD. Purdy R and Bahner RC (eds), American Society for Testing and Materials, Philadelphia, 87-103.

Adema DMM and Vink GJ, 1981. A comparative study of 1,1,2-trichloroethane, dieldrin, pentachlorophenol and 3,4-dichloroaniline for marine and fresh water organisms. Chemosphere 10, 533-554.

Allard M and Moreau G 1987. Effects of experimental acidification on a lotic macroinvertebrate community. Hydrobiologia 144, 37-49.

Allen KN, 1991. Seasonal variation of selenium in outdoor experimental stream-wetland systems. Journal of Environmental Quality 20, 865-868.

Amblard C, Couture P and Bourdier G, 1990. Effects of a pulp and paper mill effluent on the structure and metabolism of periphytic algae in experimental streams. Aquatic Toxicology 18, 137-162.

Antia NJ, McAllister CD, Parsons TR, Stevens K and Strickland JDH, 1963. further measurements of primary production using a large-volume plastic sphere. Limnol. Oceanogr. 8, 166-183.

Armitage BJ, 1980. Effects of temperature on periphyton biomass and community composition in the Browns Ferry Experimental Channels. In Giesy JP (ed.), Microcosms in Ecological Research, U.S. Department of Energy, 668-683.

Armitage BJ, Forsythe TD, Rodgers EB and Wrenn WE, 1978. Browns Ferry Biothermal Research Series I. Colonization by periphyton, zooplankton, and macroinvertebrates. US-EPA Ecological Research Series EPA-600/3-78-020. Duluth, Minnesota.

Arthur JW, 1975. Comparative toxicity of sewage-effluent disinfection to aquatic life. Report EPA-600/3-75-012. National Technical Information Service, Springfield, Virginia.

Arthur JW, 1988. Application of laboratory-derived criteria to an outdoor stream ecosystem. Intern. J. Environmental Studies 32, 97-110.

Arthur JW, Zischke JA and Ericksen GL, 1982. Effect of elevated water temperature on macroinvertebrate communities in outdoor experimental channels. Water Res. 16, 1465-1477.

Arthur JW, Zischke JA, Allen KM and Hermanutz RO, 1983. Effects of diazinon on macroinvertebrates and insect emergence in outdoor experimental channels. Aquatic Toxicology 4, 283-301.

Arthur JW, West CW, Allen KN and Hedtke SF, 1987. Seasonal toxicity of ammonia to five fish and nine invertebrate species. Bull. Environ. Contam. Toxicol. 38, 324-331.

Barton DR and Smith SM, 1984. Insects of extremely small and extremely large aquatic inhabitats. In: Resh UH and Rosenberg DM (eds.). The ecology of aquatic insects, Praeger Publ., NY.

Belanger SE, 1992. Use of experimental stream mesocosms to assess ecosystem risk: a case study with a cationic surfactant. In: Cairns JJr, Niederlehner BR and Orvose DR (eds). Predicting Ecosystem Risk. Advances in Modern Environmental Toxicology Vol. 20, Princeton Scientific Publishing Company, Inc., Princeton, NJ, 263-287.

Belanger SE, 1994. Review of experimental microcosm, mesocosm, and field tests used to evaluate the potential hazard of surfactants to aquatic life and the relation to single species data. In: Hill IR, Heimbach F, Leeuwangh P and Matthiessen P (eds.). Freshwater Field Tests for Hazard Assessment of Chemicals. Lewis Publishers, Inc., Boca Raton, Fl, 299-326.

Belanger SE and Guckert JB, 1986. Model stream ecosystem responses to alkyl sulfate and alkyl ethoxylate sulfate and comparisons with single species data. Procter & Gamble Internal Report, presented at the Mesocosm Special Session: International Workshop on the Environmental Fate and Effects of Surfactants, Veldhoven, The Netherlands, 1993, 48 pp.

Belanger SE, Farris JL, Cherry DS and Cairns JJr, 1986. Growth of Asiatic clams (*Corbicula* sp.) during and after long-term zinc exposure in field-located and laboratory artificial streams. Arch. Environ. Contam. Toxicol. 15, 427-434.

Belanger SE, Farris JL, Cherry DS and Cairns JJr, 1990. Validation of *Corbicula fluminea* growth reductions induced by copper in artificial streams and river systems. Can. J. Fish. Aquat. Sci. 47, 904-914.

Belanger SE, Davidson DH, Farris JL, Reed D and Cherry DS, 1993. Effects of cationic surfactant exposure to a bivalve mollusc in stream mesocosms. Environ. Toxicol. Chem. 12, 1789-1802.

Belanger SE, Barnum JB, Woltering DM, Bowling JW, Ventullo RM, Schermerhorn SD and Lowe RL, 1994. Algal periphyton structure and function in response to consumer chemicals in stream mesocosms. In: Graney RL, Kennedy JH and Rodgers JH (eds.), Aquatic Mesocosm Studies in Ecological Risk Assessment. Lewis Publishers, Boca Raton, Fl., 535-568.

Belanger SE, Meiers EM and Bausch RG, 1995a. Direct and indirect ecotoxicological effects of alkyl sulfate and alkyl ethoxysulfate on macroinvertebrates in stream mesocosms. Aquatic Toxicol. 33, 65-87.

Belanger SE, Rupe KL and Bausch RG, 1995b. Responses of invertebrates and fish to alkyl sulfate and alkyl ethoxylate sulfate anionic surfactants during chronic exposure. Bull. Environ. Contam. Toxicol. 55, 751-758.

Belanger SE, Rupe KL, Lowe RL, Johnson D and Pan Y, 1996. A flow-through laboratory microcosm suitable for assessing effects of surfactants on natural periphyton. Environ. Toxicol. Water Qual. 11, 65-76.

Bisson PA and Davis GE, 1976. Production of juvenile chinook salmon, *Oncorhynchus tshawytscha*, in a heated model stream. Fishery Bulletin 74, 763-774.

Borthwick PW, Clark JR, Montgomery RM, Patrick JM Jr, and Lores EM, 1985. Field confirmation of a laboratory-derived hazard assessment of the acute toxicity of fenthion to pink shrimp, *Penaeus duorarum*. In: Bahner RC and Hansen DJ (eds.), Aquatic Toxicology and Hazard Assessment: Eighth Symposium, ASTM STP 891. American Society for Testing and Materials, Philadelphia, Pennsylvania, 177-189.

Bothwell ML, Sherbot DM and Pollock CM, 1994. Ecosystem response to solar ultraviolet-B radiation: influence of trophic-level interactions. Science 265, 97-100.

Bowling JW, Giesy JP, Kania HJ and Knight RL, 1980. Large scale microcosms for assessing fates and effects of trace contaminants. In: Giesy JP (ed.), Microcosms in Ecological Research, 52. U.S. Department of Energy, 224-247.

Bowling JW, Leversee GJ, Landrum PF and Giesy JP, 1983. Acute mortality of anthracene-contaminated fish exposed to sunlight. Aquat. Toxicol. 3, 79-90.

Bowling JW, Haddock JD and Allred PM, 1984. Disposition of anthracene in the water and aufwuchs matrices of a large outdoor channel microcosm: a data set for mathematical simulation models. EPA-600/S3-84-036, Project Summary. US-EPA, Environmental Research Laboratory, Athens, GA.

Brasher AM and Ogle RS, 1993. Comparative toxicity of selenite and selenate to the amphipod *Hyallela azteca*. Arch. Environm. Contam. Toxicol. 24, 182-186.

Breneman DH and Pontasch KW, 1994. Stream microcosm toxicity tests: predicting the effects of fenvalerate on riffle insect communities. Environ. Toxicol. Chem. 13, 381-387.

Brock TCM, van den Bogaert M, Bos AR, van Breukelen SWF, Reiche R, Terwoert J, Suykerbuyk REM and Roijackers RMM, 1992. Fate and effects of the insecticide Dursban 4E in indoor Elodea-dominated and macrophyte-free freshwater model ecosystems: II. Secondary effects on community structure. Arch. Environ. Contam. Toxicol. 23, 391-409.

Burton TM and Allan JW, 1986. Influence of pH, aluminium, and organic matter on stream invertebrates. Canad. J. Fish. Aquat. Sci. 43, 1285-1289.

Call DJ and Geiger DL, 1992. Subchronic toxicities of industrial and agricultural chemicals to fathead minnows (*Pimephales promelas).* Vol. I. Center for Lake Superior Environmental Studies, Superior, Wisconsin, ISBN 0-9614968-6-X. 318 pages.

Cairns JJr, and Pratt JR, 1985. Multi-species toxicity testing using indigenous organisms- a new, cost-effective approach to ecosystem protection. Proceedings. 1985 TAPPI Environmental Conference, Atlanta, Georgia, 149-159.

Cairns JJr and Cherry DS, 1993. Freshwater multi-species test systems. In: Carlow P (ed.), Handbook of ecotoxicology, Vol. 1. Blackwell Sci. Publ., 101-116.

Cairns JJr, Niederlehner BR and Pratt JR, 1990. Evaluation of joint toxicity of chlorine and ammonia to aquatic communities. Aquat. Toxicol. 16, 87-100.

Cherry DS, Farris JL and Belanger SE, 1988. Use of ecological, biochemical, and toxicological procedures for identifying and quantifying the extent of stressful constituents in power plant effluents. Final Report to the American Electric Power Company, Biology Department and University Center for Environmental Studies. Virginia Polytechnic Institute and State University, Blacksburg, Virginia, 403 p.

Cherry DS, Farris JL and Neves RJ, 1991. Laboratory and Field Ecotoxicological Studies at the Clinch River Plant, Virginia. Final Report to the American Electric Power Company, Department of Biology, Virginia Polytechnic Institute and State University, 217 p.

Clark JR and Cripe CR, 1993. Marine and Estuarine Multi-species Test Systems. In: Calow P (ed.), Handbook of Ecotoxocology, Vol. 1. Blackwell Scientific Publications, London Edinburgh Boston.

Clark JR and Noles JL, 1994. Contaminant effects in marine/estuarine systems: Field studies and scaled simulations. In: Garney RL, Kennedy JH and Rodgers JH (eds.), Aquatic Mesocosm Studies in Ecological Risk Assessment. Lewis Publishers, Boca Raton Ann Arbor London Tokyo, 47-60.

Clark JR, Cherry DS and Cairns JJr, 1982. Food quality of Aufwuchs from artificial streams receiving low levels of perturbations. Water Resources Bull. 18, 761-767.

Clark JR, Borthwick PW, Goodman LR, Patrick JM Jr, Lores EM and Moore JC, 1987. Comparison of laboratory toxicity test results with responses of caged estuarine animals exposed to fenthion in the field. Environ. Toxicol. Chem. 6, 151-160.

Clements WH, Cherry DS and Cairns JJr, 1988. Structural alterations in aquatic insect communities exposed to copper in laboratory streams. Environ. Toxicol. Chem. 7, 715-722.

Clements WH, Farris JL, Cherry DS and Cairns JJr, 1989. The influence of water quality on macroinvertebrate community responses to copper in outdoor experimental streams. Aquat. Toxicol. 14, 249-262.

Cole LK and Metcalf RL, 1980. Environmental destinies of insecticides, herbicides and fungicides in the plants, animals, soil, air and water of homologous microcosms. In: Giesy JPJr (ed.), Microcosms in ecological research. US Dept. Energy. Technical Information Service, pp. 971-1007.

Crossland NO, 1988a. A method for evaluating effects of toxic chemicals on the productivity of freshwater ecosystems. Ecotox. and Environ. Safety 16, 279-292.

Crossland NO 1988b. The design of experimental streams for studying biological effects of chemicals and effluents in receiving waters. External Report SBER 88-009, Shell Research Limited, Sittingbourne Research Centre, Kent, England.

Crossland NO, Heimbach F, Hill IR, Boudou A, Leeuwangh P, Matthiessen P and Persoone G, 1992a. Summary and recommendations of the European Workshop on Freshwater Field Tests held at Potsdam 25-26 June 1992, SETAC Publication.

Crossland NO, Mitchell GC and Dorn PB, 1992b. Use of outdoor artificial streams to determine threshold toxicity concentrations for a petrochemical effluent. Environ. Toxicol. Chem. 11, 49-59.

Crouse MR, Callahan CA, Malueg KW and Dominguez SE, 1981. Effects of fine sediments on growth of juvenile coho salmon in laboratory streams. Transactions of the American Fisheries Society 110, 281-286.

Cuffney TF, Hart DD, Wolbach KC, Wallace JB, Luthgart GJ and Smith-Cuffney FL, 1990. Assessment of community and ecosystem level effects in lotic environments: the role of mesocosm and field studies. Proceedings, North American Benthological Society Technical Information Workshop "Experimental Ecosystems: Applications to Ecotoxicology". Blacksburg, Virginia, 40.

de Kozlowski SJ and Bunting II DL, 1981. A laboratory study on the thermal tolerance of four southeastern stream insect species (Trichoptera, Ephemeroptera). Hydrobiologia 79, 141-145.

Deutsch WG, Webber EC, Bayne DR and Reed CW, 1992. Effects of the largemouth bass stocking rates on fish populations in aquatic mesocosms used for pesticide research. Environ. Toxcol. Chem. 11, 5-10.

Dionne E, 1978. Chronic toxicity of CGA-24705 to the fathead minnow (*Pimephales promelas*). EG & G Bionomics, Aquatic Tox. Labs. Inc., Madera, California, USA. Ciba-Geigy internal Report #BW-78-11-341, CG 25705/193.

Dorn PB, Rodgers JHJr, Dubey ST, Gillespie WB and Figueroa AR, 1994. An assessment of the ecological effects of a C9-11 linear alcohol ethoxylate surfactant in stream mesocosm experiments. Unpublished report. Shell Development Company, Westhollow Technology Center, Houston, Texas, and the University of Mississippi, Department of Biology. 26 p.

Dorn PB, Rodgers JHJr, Dubey ST, Gillespie WB and Figueroa AR, 1996a. Assessing the effects of a C_{14-15} linear alcohol ethoxylate surfactant in stream mesocosms. Ecotoxicol. Environ. Safety 34, 196-204.

Dorn PB, Tattersfield L, Holt M and Dubey ST, 1996b. Development of effects data for alcohol ethoxylate surfactants using stream mesocosms. Proceedings of the CESIO 4th World Surfactants Congress, June 3-7, 1996. Barcelona, Spain, pp. 197-212.

Dortland RJ, 1980. Toxicological evaluation of parathion and azinphosmethyl in freshwater model ecosystems. Agricultural Research Report 898, Centre for Agricultural Publishing and Documentation, Wageningen, The Netherlands.

Eaton J, Arthur J, Hermanutz R, Kiefer R, Mueller L, Anderson R, Erickson R, Nordling B, Rogers J and Pritchard H, 1985. Biological effects of continuous and intermittent dosing of outdoor experimental streams with chlorpyrifos. In: Bahner RC and Hansen DJ (eds), Aquatic Toxicology and Hazard Assessment: Eighth Symposium, ASTM STP 891 American Society for Testing and Materials, Philadelphia, Pennsylvania, 85.

ECETOC, 1993a. Environmental hazard assessment of substances. Technical Report No. 51.

ECETOC, 1993b. Aquatic toxicity data evaluation. Technical Report No. 56.

ECETOC, 1994a. Assessment of non-occupational exposure to chemicals. Technical Report No. 58.

ECETOC, 1994b. Environmental exposure assessment. Technical Report No. 61.

ECETOC, 1996. The role of bioaccumulation in environmental risk assessment: The aquatic environment and related food webs. Technical Report No. 67.

EEC, 1996. Technical guidance documents in support of the Commission Directive 93/67/EEC on risk assessment for new notified substances and the Commission Regulation (EC) 1488/94 on risk assessment for existing substances, Brussels.

Eichenberger E, Schlatter F, Weilenmann H and Wuhrmann K, 1981. Toxic and eutrophicating effects of Co, Cu and Zn on algal benthic communities in rivers. Verh. Internat. Verein. Limnol. 21, 1131-1134.

Fairchild JF, Boyle T, English WR and Rabeni CF, 1987. Effects of sediment and contaminated sediment on structural and functional components of experimental stream ecosystems. Water, Air and Soil Pollution 36, 271-293.

Fairchild JF, Dwyer FJ, La Point TW, Burch SA and Ingersoll CG, 1993. Evaluation of a laboratory-generated NOEC for linear alkylbenzene sulfonate in outdoor experimental streams. Environ. Toxicol. Chem. 12, 1763-1775.

Farris JL, 1986. Cellulolytic responses to heavy metal accumulation in *Corbicula fluminea* and *Mudalia dilatata*. Ph.D. Thesis, Virginia Polytechnic Institute and State University, Blacksburg, VA .

Farris JL, Van Hassel JH, Belanger SE, Cherry DS and Cairns JJr, 1988. Application of cellulolytic activity of Asiatic clams (*Corbicula* Sp.) to in-stream monitoring of power plant effluents. Environ. Toxicol. Chem. 7, 701-713.

Farris JL, Belanger SE, Cherry DS and Cairns JJr, 1989. Cellulolytic activity as a novel approach to assess long-term zinc stress to *Corbicula*. Water Res. 23, 1275-1284.

Farris JL, Cherry DS and Neves RJ, 1991. Validation of copper concentrations in laboratory testing for site-specific copper criteria in the Clinch River. Final Report to the American Electric Power Company, Department of Biology, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, 171 p.

Farris JL, Grudzien JL, Belanger SE, Cherry DS and Cairns JJr, 1994. Molluscan cellulolytic activity responses to zinc exposure in laboratory and field stream comparisons. Hydrobiologia 287, 161-178.

Feijtel TCJ and van de Plassche EJ, 1995. Environmental risk characterisation of 4 major surfactants in the Netherlands. RIVM/NVZ report 679101 025. September 1995, Bilthoven, The Netherlands.

Ferens MC, 1974. The impact of mercuric ions on benthos and periphyton of artificial streams. Ph.D. Dissertation. University of Georgia, Athens, Georgia.

Geckler JR, Horning WB, Neiheisel TM, Pickering QH, Robinson EL and Stephan CE, 1976. Validity of laboratory tests for predicting copper toxicity in streams. EPA-600/3-76-116. Technical Report, United States Environmental Protection Agency.

Genter RB and Amyot DJ, 1994. Freshwater benthic algal population and community changes to acidity and aluminium-acid mixtures in artificial streams. Environ. Toxicol. Chem. 13, 369-380.

Genter RB, Cherry DS, Smith EP and Cairns JJr, 1987. Algal-periphyton population and community changes from zinc stress in stream mesocosms. Hydrobiologia 153, 261-275.

Genter RB, Colwell FS, Pratt J R, Cherry DS and Cairns JJr, 1988a. Changes in epilithic communities due to individual and combined treatments of zinc and snail grazing in stream mesocosms. Toxicol. Ind. Health 4, 185-201.

Genter RB, Cherry DS, Smith EP and Cairns JJr, 1988b. Attached-algal abundance altered by individual and combined treatments of zinc and pH. Environ. Toxicol. Chem. 7, 723-733.

Gerhart DZ, Anderson SM and Richter J, 1977. Toxicity bioassays with periphyton communities: design of experimental streams. Water Res. 11, 567-570.

Giddings JM, 1986. A microcosm procedure for determining safe levels of chemical exposure in shallow water communities. In: Cairns JJr. (ed.), Community Toxicity Testing, ASTM STP 920, American Society for Testing and Materials, Philadelphia, Pennsylvania.

Giesy JP, Kania HJ, Bowling JW, Knight RL, Mashburn S and Clarkin S, 1979. Fate and biological effects of cadmium introduced into channel microcosms. EPA-600/3-79-039. US Environmental Protection Agency, Athens, Georgia, 157 p.

Giesy JP, Bowling JW, Kania HJ, Knight RL and Mashburn S, 1981. Fates of cadmium introduced into channels microcosm. Environ. Internat. 5, 159-175.

Giesy JP, Bartell SM, Landrum PF, Leversee GJ and Bowling JW, 1983a. Fates and biological effects of polycyclic aromatic hydrocarbons in aquatic systems. EPA-600/S3-83-053, Project Summary. United States Environmental Protection Agency, Athens, Georgia, 5 p.

Giesy JP, Bartell SM, Landrum PF, Leversee GJ, Bowling JW, Bruno MG, Fannin TE, Gerould S, Haddock JD, LaGory K, Oris JT and Spacie A, 1983b. Fates and biological effects of polycyclic aromatic hydrocarbons in aquatic systems. IAG EPA-78-D-X0290. Internal Report. United States Environmental Protection Agency, Athens, Georgia, 226 p.

Gillespie WB, Rodgers JH Jr and Crossland NO, 1996a. Effects of a non-ionic surfactant (C14-15AE7) on aquatic invertebrates in outdoor stream mesocosms. Environmental Toxicology and Chemistry 15, 1418-1422.

Gillespie WB Jr, Wong DCL, Dorn PB and Rodgers JH Jr,. 1996b. Chronic toxicity of a homologous series of alcohol ethoxylate surfactants to *Daphnia magna*. Report of 10 April 1996 of the University of Mississippi and Shell Development Company, Houston, Texas, USA, shared with RIVM.

Grade R, 1993. Report on the growth inhibition test of GS 13529 tech. to green algae (*Scenedesmus subspicatus*). Rep. No. 928431. Ciba-Geigy Ltd., Basle.

Graney RL, Kennedy JH and Rodgers JH (eds), 1994. Aquatic Mesocosm Studies in Ecological Risk Assessment. CRC Press, Boca Raton, Florida.

Grice GD and Menzel DW, 1978. Controlled ecosystem pollution experiment: effect of mercury on enclosed water columns. VIII. Summary of results. Marine Science Communications 4, 23-31.

Grice GD and Reeve MR (eds.), 1982. Marine mesocosms. Biological and chemical research in experimental ecosystems. Springer-Verlag, New York Heidelberg Berlin.

Guckert JB, Walker DD, and Belanger SE, 1996. Environmental chemistry for a surfactant ecotoxicology study supports rapid degradation of C_{12}-alkyl sulfate in a continuous-flow stream mesocosm. Environ. Toxicol. Chem. 15, 262-269.

Guhl W, 1987. Beitrag zur biologischen Bewertung von Umweltchemikalien mit Laborökomodellen. I. Konzeption der Oberflächengewässermodelle und Substanzbewertung. Z. Angew. Zool. 74, 385-409.

Guhl W, 1994. Modèles écologiques aquatiques pour une évaluation améliorée du risque de l'environment. Colloque Européen „Le recours biologique pour la surveillance en continu de la qualité des eaux. Nancy, pp. 125-131.

Guhl W, 1996. Fettalkoholsulfate im Flußsimulationsmodell - eine umweltrelevante Tensidabsicherung. Internal report No. R-9600623. Henkel KGaA, Düsseldorf.

Guhl W and Gode P, 1989. Correlations between lethal and chronic/biocenotic effect concentrations of surfactants. Tenside 26, 5-8.

Hall TJ, Haley RK and LaFleur LL, 1991. Effects of biologically treated bleached kraft mill effluent on cold water stream productivity in experimental stream channels. Environ. Toxicol. Chem. 10, 1051-1060.

Hall TJ, Haley RK, Borton DL, Walsh AH and Wole RE, 1992. Histopathology of rainbow trout (*Oncorhynchus kisutch*) after long-term exposure to biologically treated bleached kraft mill effluent in experimental stream channels. Canadian J. Fish. Aquat. Sci. 49, 939-944.

Hansen SR and Garton RR, 1982a. The effects of diflubenzuron on a complex laboratory stream community. Arch. Environm. Contam. Toxicol. 11, 1-10.

Hansen SR and Garton RR, 1982b. Ability of standard toxicity tests to predict the effects of the insecticide diflubenzuron on laboratory stream communities. Canad. J. Fish. Aquat. Sci. 39, 1273-1288.

Hatakeyama S, Fukushima S, Kasai F and Shiraishi H, 1994. Assessment of herbicide effects on algal production in the Kokai River (Japan) using a model stream and *Selenastrum* bioassay. Ecotoxicol. 3, 143-156.

Havens KE, 1994. Structural and functional responses of a freshwater plankton community to acute copper stress. Environ. Pollut. 86, 259-266.

Hedtke SF, 1984. Structure and function of copper-stressed aquatic microcosms. Aquat. Toxicol. 5, 227-244.

Hedtke SF and Arthur JW, 1985. Evaluation of a site-specific water quality criterion for pentachlorophenol using outdoor experimental streams. In: Cardwell RD, Purdy R and Bahner RC (eds), Aquatic Toxicology and Hazard Assessment: Seventh Symposium. ASTM STP 854, American Society for Testing and Materials, Philadelphia, Pennsylvania, pp. 551-564.

Hedtke SF, West CW, Allen KN, Norberg-King TJ and Mount DI, 1986. Toxicity of pentachlorophenol to aquatic organisms under naturally varying and controlled environmental conditions. Environ. Toxicol. Chem. 5, 531-542.

Hedtke SF, Hermanutz RO, Allen KN and Roush TH, 1989. Evaluation of the water quality criteria for selenium. Internal Report, US Environmental Protection Agency, Duluth, Minnesota, 35 p.

Heimbach F, Berndt J and Pflueger W, 1994. Fate and biological effects of an herbicide on two artificial pond ecosystems of different size. In: Graney RL, Kennedy JH and Rodgers JH (eds.), Aquatic Mesocosm Studies in Ecological Risk Assessment. CRC Press, Boca Raton, Florida, pp. 339-344.

Hermanutz RO, 1992. Malformation of the fathead minnow (*Pimephales promelas*) in an ecosystem with elevated selenium concentrations. Bull. Environ. Contam. Toxicol. 49, 290-294.

Hermanutz RO, Hedtke SF, Arthur JW, Andrew RW and Allen KN, 1987. Ammonia effects on microinvertebrates and fish in outdoor experimental streams. Environmental Pollution 47, 249-283.

Hermanutz RO, Allen KN and Hedtke SF, 1990. Toxicity and fate of total residual chlorine in outdoor experimental streams. In: Jolley RL, Condie LW, Johnson DJ, Katz S, Minear RA, Mattice JS and Jacobs VA (eds). Water Chlorination: Chemistry, Environmental Impact and Health Effects. Vol. 6, Lewis Publishers, Inc., Chelsea, Michigan, pp. 463-477.

Hermanutz RO, Allen KN, Roush TH and Hedtke SF, 1992. Effects of elevated selenium concentrations on bluegills (*Lepomis macrochirus*) in outdoor experimental streams. Environ. Toxicol. Chem. 11, 217-224.

Hitz HR, 1981. Report on the growth inhibition of algae (*Scenedesmus subspicatus*) by G 24 480. Internal Report No. 810796. Ciba-Geigy Ltd., Basle.

Hill IR, Heimbach F, Leeuwangh P and Matthiessen P (eds.), 1994. Freshwater Field Tests for Hazard Assessment of Chemicals. Lewis Publishers. Inc., Boca Raton, FL, USA.

Hoberg JR, 1995. Metolachlor Technical - 5-day toxicity to the freshwater green alga, *Selenastrum capricornutum*, using acetone as a carrier solvent. Springborn Laboratories Inc. Mass. USA; Rep.No. 94-7-5382. CG 24705/2477. Ciba-Geigy Ltd., Basle, CH.

Holt MS and Mitchell GC, 1994. The fate and effects of linear alkylbenzene sulphonate (LAS) in outdoor artificial streams - developing an approach. Proceedings of the International Seminar on Surfactants/Detergents. 10-13 November 1994, Xian, China., 31-38.

Howey TW and Arthur JW, 1978. Seasonal changes in zooplankton abundance in an ambient and heated experimental stream. American Zoologist 18, 58-70.

Huber W, 1995a. Effects of metolachlor in aquatic outdoor microcosms. Report CG 24705/2460, Ciba-Geigy Ltd., Basle.

Huber W, 1995b. Effects of terbutylazine in aquatic outdoor microcosms. Report 1322, Ciba-Geigy Ltd., Basle.

Huber W, Zieris F-J, Feind D. and Neugebaur K, 1987. Ecotoxicological evaluation of environmental pollutions with the help of aquatic model ecosystems. - Final Report on a project sponsored by "Bundesministerium für Forschung und Technologie" (Germany), Ref. No. 03-7314-0. 80 p.

Kania HJ, Knight RL and Beyers RJ, 1976. Fate and biological effects of mercury introduced into artificial streams. EPA-600/3-76-060. Final Report. US Environmental Protection Agency, Athens, Georgia, 129 p.

Kiffney PM and Clements WH, 1994a. Effects of heavy metals on a Rocky Mountain stream macroinvertebrate assemblage in experimental streams. Colorado State University, Department of Fishery and Wildlife Biology, Fort Collins, Co, 33 p.

Kiffney PM and Clements WH, 1994b. Structural responses of benthic invertebrate communities from different stream orders to zinc. Environ. Toxicol. Chem. 13, 389-397.

Kline ER, Figueroa RA, Rodgers JH Jr, and Dorn PB, 1996. Effects of a non-ionic surfactant (C14-15AE-7) on fish survival, growth and reproduction in the laboratory and in outdoor stream mesocosms. Environ. Toxicol. Chem. 15, 997-1002.

Kosinski RJ, 1984. The effect of terrestrial herbicides on the community structure of stream periphyton. Environmental Pollution, Series A, 36,165-189.

Kosinski RJ and Merkle MG, 1984. The effect of four terrestrial herbicides on the productivity of artificial stream algal communities. J. Environ. Qual. 13, 75-82.

Kühn R, Pattard M, Parnak K-D and Winter A, 1989. Results of the harmful effects of water pollutants to *Daphnia magna* in the 21 day reproduction test. - Water Res. 23, 501-510.

Kühn R. and Pattard M, 1990. Results of the harmful effects of water pollutants to green algae (*Scenedesmus suspicatus*) in the cell multiplication inhibition test. Water Res. 24, 31-38.

Kuiper J, 1977. Development of North Sea coastal plankton communities in separate plastic bags under identical conditions. Marine Biology 44, 97-107.

Kuiper J, 1982a. Ecotoxicological experiments with marine plankton communities in plastic bags. In: Grice GD and Reeve MR (eds.) Marine Microcosms, Biological and Chemical Research in Experimental Ecosystems. Springer-Verlag, New York Heidelberg Berlin.

Kuiper J, 1982b. Netherlands organisation for applied scientific research for CEC. TNO Report No. CL82/01, cited in: Krijgsheld KR and van der Gen A, Chemosphere 15, 825-860, 1986.

Kuiper J and Hanstveit AO, 1984a. Fate and Effects of 4-chlorophenol and 2,4-dichlorophenol in marine plankton communities in experimental enclosures. Ecotoxicol. Environ. Safety 8, 15-33.

Kuiper J and Hanstveit AO, 1984b. Fate and effects of 3,4-dichloroaniline (DCA) in marine plankton communities in experimental enclosures. Ecotoxicol. Environ. Safety 8, 34-54.

Landrum PF, Bartell SM, Giesy JP, Leversee GJ, Bowling JW, Haddock J, LaGory K, Gerould S and Bruno M, 1984. Fate of anthracene in an artificial stream: a case study. Ecotoxicol. Environm. Safety. 8, 183-201.

Larson RJ and Maki AW, 1982. Effect of linear alkylbenzene sulfonate on the structure and function of microbial communities in model ecosystems. In: Pearson JG, Foster RB and Bishop WE (eds), Aquatic Toxicology and Hazard Assessment: Fifth Conference, STP 766, ASTM, Philadelphia, PA. 120-136.

Lauth JR, Cherry DS and Cairns JJr, 1990. A single reconstituted water formulation for the culture and toxicity testing of algae, invertebrates and fish. Environmental Auditor 1, 109-219.

Leffler JW, 1981. Aquatic microcosms and stress criteria for assessing environmental impact of organic chemicals. Office of Pesticide and Toxic Substances. US-EPA, Washington, DC.

Leland HV and Kent E, 1981. Effects of copper on microfaunal species composition in a Sierra Nevada, California, stream. Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 21, 819-829.

Leland HV and Carter JL, 1985. Effects of copper on production of periphyton, nitrogen fixation and processing of leaf litter in a Sierra Nevada, California, stream. Freshwater Biology 15, 155-173.

Leland HV, Fend SV, Dudley TL and Carter JL, 1989. Effects of copper on species composition of benthic insects in a Sierra Nevada, California, stream. Freshwater Biology 21, 163-179.

Lewis MA, Taylor MJ and Larson RJ, 1986. Structural and functional response of natural phytoplankton and periphyton communities to a cationic surfactant with considerations on environmental fate. In: Cairns JJr (ed), Community Toxicity Testing, ASTM STP 920, American Society for Testing and Materials, Philadelphia, Pennsylvania, 241-268.

Lewis MA, Pittinger CA, Davidson DH and Ritchie CJ, 1993. In-situ response of natural periphyton to an anionic surfactant and an environmental risk assessment for phytotoxic effects. Environ. Toxicol. Chem. 12, 1803-1812.

Liber K, Kaushik NK, Solomon KR and Carey JH, 1992. Experimental designs for aquatic mesocosm studies: a comparison of the "Anova" and "Regression" design for assessing the impact of tetrachlorophenol on zooplankton populations in limnocorrals. Environm. Toxicol. Chem. 11, 61-77.

Lizotte RE Jr, Wong DCL, Dorn PB and Rodgers JH Jr, 1996. Chronic toxicity of a homologous series of alcohol ethoxylate surfactants to fathead minnow. Report of 10 April 1996 of the University of Mississippi and Shell Development Company, Houston, Texas, USA, shared with RIVM.

Lynch TR, Johnson HE and Adams WJ, 1982. The fate of atrazine and a hexachlorobiphenyl isomer in naturally-derived model stream ecosystems. Environ. Toxicol. Chem. 1, 179-192.

Lynch TR, Johnson HE and Adams WJ, 1985. Impact of atrazine and hexachlorobiphenyl on the structure and function of model stream ecosystems. Environ. Toxicol. Chem. 4, 399-413.

Macek K, Weber EC, Hogan JW and Holtz DD, 1972. Toxicity of the insecticide Dursban to fish and aquatic invertebrates in ponds. Trans. Am. Fish. Soc. 3, 420-427.

Maki AW 1979. Correlations between *Daphnia magna* and fathead minnow (*Pimephalis promelas*) chronic toxicity values for several classes of test substances. J. Fish. Res. Board Can. 36, 411-421.

Maki AW 1980. A laboratory model ecosystem approach to environmental fate and effects studies. Research Report, Procter and Gamble Company, Cincinnati, Ohio, 43 p.

Maki AW and Johnson HE, 1976. Evaluation of a toxicant on the metabolism of model stream communities. J. Fish. Res. Board Can. 33, 2740-2746.

Maki AW and Johnson HE, 1977. Kinetics of lampricide (TFM, 3-trifluoromethyl-4-nitrophenol) residues in model stream communities. J. Fish. Res. Board Can. 34, 276-281.

Maki AW, Geissel L and Johnson H E, 1975. Comparative toxicity of larval lampricide (TFM: 3-trifluoromethyl-4-nitrophenol) to selected benthic macroinvertebrates. J. Fish. Res. Board Can. 32, 1455-1459.

Matida Y, Kimura S, Kumada H and Yokote M, 1975. Effects of some herbicides applied in the forest to the fresh water fishes and other aquatic organisms: II. Effects of sodium chlorate and ammonium sulfamate to the aquatic organisms in the artificial stream. Bull. Freshwater Fish. Res. Lab. (Tokyo) 25, 55.

Mauch E, 1988. Die Auswirkungen des Chemie-Unfalles bei Sandoz in Basel im November 1986 auf die Bodenbesiedlung des Rheins bei Karlsruhe. GWF Wasser-Abwasser 129, 168-171.

McAllister CD, Parsons TR, Stephens K and Strickland JDH, 1961. Measurements of primary production in coastal sea water using a large-volume plastic sphere. Limnology and Oceanography VI. 237-258.

McCarthy JF and Jimenez BD, 1985. Reduction in bioavailability to bluegills of polycyclic aromatic hydrocarbons bound to dissolved humic material. Environ. Toxicol. Chem. 4, 511-521.

McCormick JH, Stokes GN and Portele GJ, 1980. Environmental acidification impact detection by examination of mature fish ovaries. 975. Canadian Technical Report of Fisheries and Aquatic Sciences, United States Environmental Protection Agency, Duluth, Minnesota.

McCormick PV, Pratt JR and Cairns JJr, 1986. Effect of 3-trifluoromethyl-4-nitrophenol on the structure and function of protozoan communities established on artificial substrates. In: Cairns JJr (ed.), Community Toxicity Testing, ASTM STP 920, American Society for Testing and Materials, Philadelphia, Pennsylvania, 224-240.

McCormick JH, Stokes GN and Hermanutz RO, 1989. Oocyte atresia and reproductive success in fathead minnows (*Pimephales promelas*) exposed to acidified hardwater environments. Arch. Environ. Contam. Toxicol. 18, 207-214.

McCormick PV, Belanger SE, Niederlehner BR and Cairns JJr, in Press. Evaluating the hazard of dodecyl alkyl sulfate to natural ecosystems using protozoan communities. Ecotoxicology.

Menzel DW and Case J, 1977. Concept and design: Controlled ecosystem pollution experiment. Bull. Marine Sci. 27, 1-7.

MERS (Monticello Ecological Research Station), 1988. The impact of chlorine/ammonia on ecosystem structure and function in experimental streams. Report to the Office of Water. US Environmental Protection Agency, Duluth, Minnesota, 44 pp.

Mitchell GC, Bennett D and Pearson N, 1993. Effects of lindane on macroinvertebrates and periphyton in outdoor artificial streams. Ecotoxicol. Environ. Safety 25, 90-102.

Miura T and Takahashi RM, 1974. Insect developmental inhibitors. Effects of candidate mosquito control agents on nontarget aquatic organisms. Environ. Entomol. 3, 631-636.

Moorhead DL and Kosinksi RJ, 1986. Effect of atrazine on the productivity of artificial stream algal communities. Bull. Environ. Contam. Toxicol. 37, 330-336.

Mundie JH, Simpson KS, and Perrin CJ, 1991. Responses of stream periphyton and benthic insects to increases in dissolved inorganic phosphorus in a mesocosm. Can. J. Fish. Aquat. Sci. 48, 2061-2072.

NCASI, 1983. Effects of biologically stabilized bleached kraft effluent on warm water stream productivity in experimental streams - third progress report. Technical Bulletin 414, National Council of the Paper Industry for Air and Stream Improvement, New York, NY.

NCASI, 1985. Effects of biologically treated bleached kraft mill effluent on cold water stream productivity in experimental stream channels - fourth progress report. Technical Bulletin 474, National Council of the Paper Industry for Air and Stream Improvement, New York, NY.

NCASI, 1991. Observations on the bioaccumulation of 2,3,7,8-TCDD and 2,3,7,8-TCDF in channel catfish and largemouth bass and their survival or growth during exposure to biologically treated kraft mill effluent in experimental streams. Technical Bulletin 611. National Council of the Paper Industry for Air and Stream Improvement, New York, NY.

NCASI, 1993. Aquatic community effects of biologically treated bleached kraft mill effluent before and after conversion to increased chlorine dioxide substitution: results from an experimental streams study. Technical Bulletin 653. National Council of the Paper Industry for Air and Stream Improvement, New York, NY.

Nagel R, Bresch H, Caspers N, Hansen PD, Markert M, Munk R, Scholz N and ter Höfte BB, 1991. Effect of 3,4-dichoroaniline on the early life stages of the zebrafish (*Brachydanio rerio*): Results of a comparative laboratory study. Ecotoxicology and Environmental Safety 21, 157-164.

Newman RM and Perry JA, 1989. The combined effects of chlorine and ammonia on litter breakdown in outdoor experimental streams. Hydrobiologia 184, 69-78.

Newman RM, Perry JA, Tam E and Crawford RL, 1987. Effects of chlorine exposure on litter processing in outdoor experimental streams. Freshwater Biol. 18, 415-428.

Niederlehner BR and Cairns JJr, 1993. Effects of previous zinc exposure on pH tolerance of periphyton communities. Environ. Toxicol. Chem. 12, 743-753.

Niederlehner BR, Pontasch KW, Pratt JR and Cairns JJr, 1990. Field evaluation of predictions of environmental effects from a multi-species microcosm toxicity test. Arch. Environ. Contam. Toxicol. 19, 62-71.

Nordlie KJ and Arthur JW, 1981. Effect of elevated water temperature on insect emergence in outdoor experimental channels. Environmental Pollution (Series A) 25, 53-65.

Odum EP, 1984. The Mesocosm. BioScience 34, 558-562.

Odum HT, Siler WL, Beyers RJ and Armstrong N, 1963. Experiments with engineering of marine ecosystems. Contrib. Marine Sci., Univ. Texas, 9, 373-403.

OECD, 1992. Provisional Guidance for Initial Hazard Assessment of HPV Chemicals with a Full SIDS. Initial Assessment of Aquatic Effects Document No. 92,126/13.08.92

Ogle RS and Wright AW, 1996. Selenium bioaccumulation in aquatic ecosystems: 1. Effects of sulfate on the uptake and toxicity of selenate in *Daphnia magna*. Arch. Environm. Contam. Toxicol. 30, 274-279.

Oviatt CA, Buckley B and Nixon S, 1981. Annual phytoplankton metabolism in Naragansett Bay calculated from survey field measurements and microcosm observations. Estuaries 4, 167-175.

Perrin CJ, Wilkes B and Richardson JS, 1992. Stream periphyton and benthic insect responses to additions of treated acid mine drainage in a continuous-flow on-site mesocosm. Environ. Toxicol. Chem. 11, 1513-1525.

Pignatello JJ, Johnson LK, Martinson MM, Carlson RE and Crawford RL, 1986. Response of the microflora in outdoor experimental streams to pentachlorophenol: environmental factors. Can. J. Microbiol. 32, 38-46.

Planas D, Lapierre L, Moreau G and Allard M, 1989. Structural organization and species composition of a lotic periphyton community in response to experimental acidification. Can. J. Fish. Aquat. Sci. 46, 827-835.

Pontasch KW and Cairns JJr., 1991. Multispecies toxicity tests using indigenous organisms: predicting the effects of complex effluents in streams. Arch. Environ. Contam. Toxicol. 20, 103-112.

Pontasch KW, Niederlehner BR and Cairns JJr, 1989a. Comparisons of single-species, microcosm and field responses to a complex effluent. Environ. Toxicol. Chem. 8, 521-532.

Pontasch KW, Smith EP and Cairns JJr, 1989b. Diversity indices, community comparison indices and canonical discriminant analysis: interpreting the results of multispecies toxicity tests. Water Res. 23, 1229-1238.

Pratt JR and Bowers NJ, 1990. Effect of selenium on microbial communities in laboratory microcosms and outdoor streams. Toxicity Assessment 5, 293-307.

Pratt JR, Niederlehner BR, Bowers NJ, and Cairns JJr, 1987a. Effects of zinc on freshwater microbial communities. In: Lindberg SE and Hutchinson TC (eds), Proceedings, International Conference on Heavy Metals in the Environment, New Orleans, Vol. 2. CEP Consultants Ltd, Edinburgh, pp. 324-326.

Pratt JR, Niederlehner BR, Bowers N and Cairns JJr, 1987b. Prediction of permissible concentrations of copper from microcosm toxicity tests. Toxicity Assessment 2, 417-436.

Pratt JR, Bowers NJ, Niederlehner BR and Cairns JJr, 1988a. Effects of atrazine on freshwater microbial communities. Arch. Environ. Contam. Toxicol. 17, 449-457.

Pratt JR, Bowers NJ, Niederlehner BR and Cairns JJr, 1988b. Effects of chlorine on microbial communities in naturally derived microcosms. Environ. Toxicol. Chem. 7, 679-687.

Pratt JR, Bowers NJ, Niederlehner BR and Cairns JJr, 1989. Response of laboratory ecosystems to environmental stress: effect of phenol. Toxicity Assessment 4, 161-174.

Pusey BJ, Arthington AH and McLean J, 1994a. The effects of a pulsed application of chlorpyrifos on macroinvertebrate communities in an outdoor artificial stream system. Ecotoxicol. Environ. Safety 27, 221-250.

Pusey BJ, Arthington AH and Flanders TJ, 1994b. An outdoor replicated artificial stream system: design, operating conditions and initial invertebrate colonization. Ecotoxicol. Environ. Safety 27, 177-191.

Putt AE, 1995. Metolachlor Technical - The chronic toxicity to *Daphnia magna* under flow-through conditions. Springborn Laboratories Inc. Mass, USA; Rep.No. 95-8-6061, CG 24705/2534. Ciba-Geigy Ltd., Basle.

Ritter A, 1990. Terbuthylazin technical GS 13529: 21-day prolonged toxicity study in the rainbow trout under flow-through conditions. RCC Umweltchemie AG, Itingen, CH. Rep.No. 227248. CG 13529/919. Ciba-Geigy Ltd., Basle.

Rodgers EB, 1980. Effects of elevated temperatures on macroinvertebrate populations in the Browns Ferry experimental ecosystems. In: Giesy JP (ed.), Microcosms in Ecological Research, U.S. Department of Energy, pp. 684-702.

Rodgers JHJr, Clark JR, Dickson KL and Cairns JJr, 1980. Nontaxonomic analyses of structure and function of aufwuchs communities in lotic microcosms. In: Giesy JP (ed.), Microcosms in Ecological Research, U.S. Department of Energy, pp. 625-644.

Rodgers JH Jr, Crossland NO, Kline ER, Gillespie WB Jr, Figueroa RA and Dorn PB, 1996. Design and construction of model stream mesocosms. Ecotoxicol. Environ. Safety 33, 30-37.

Rosenberg DM and Resh UH (eds.), 1992. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, NY.

Rufli H, 1989. Report on the Daphnia, reproduction test with GS 13529 technical. Rep. No. 891104. Ciba-Geigy Ltd, Basle.

Russell PP, Horne AJ and Thomas JF, 1981. Application of laboratory scale model streams toward assessing effluent impacts in freshwater lotic environments. In: Bates JM and Weber CI (eds), Ecological Assessments of Effluent Impacts on Communities of Indigenous Aquatic Organisms. ASTM STP 730, American Society for testing and Materials, Philadelphia, Pennsylvania, 32-48.

Sauter S, Buxton KS, Macek KJ and Petrocelli SR, 1976. Effects of exposure to heavy metals on selected freshwater fish. - Toxicity of copper, cadmium, chromium and lead to eggs and fry of seven fish species. Research Rep. No. EPA-600/3-76/105. Environmental Protection Technology Series, US-EPA, National Technical Information Service, Springfield, VA. 75.

Schäfer H, Hettler H, Fritsche U, Pitzen G, Röderer G and Wenzel A, 1994. Biotests using unicellular algae and ciliates for predicting long-term effects of toxicants. - Ecotox. Environ. Safety 27, 64-81.

Schindler DW, Mills KH, Malley DF, Findlay DL, Shearer JA, Davies IJ, Turner MA, Linsey GA and Cruikshank DR, 1985. Long-term ecosystem stress: the effects of years of experimental acidification on a small lake. Science 228, 1395-1401.

Scholz N and Müller FJ, 1992. The riverine biocoenosis model (aquatic stair case model): A test system for determining the ecotoxicity and biodegradation under reality-approximate riverine conditions. Chemosphere 25, 563-579.

Schultz R and Hermanutz R, 1990. Transfer of toxic concentrations of selenium from parent to progeny in the fathead minnow (*Pimephales promelas*). Bull. Environ. Contam. Toxicol. 45, 568-573.

Schwab BS, Maruscik DA, Ventullo RM and Palmisano AC, 1992. Adaptation of periphytic communities in model streams to a quaternary ammonium surfactant. Environ. Toxicol. Chem. 11, 1169-1177.

SETAC Europe, 1991. Guidance document on testing procedures for pesticides in freshwater mesocosms, from the workshop "A Meeting of Experts on Guidelines for Static Field Mesocosm Tests", Huntingdon , UK, 46 p.

SETAC-RESOLVE, 1992. Proceedings of a workshop on aquatic microcosms for ecological assessment of pesticides (Wintergreen, Virginia, USA, October 1991). SETAC Foundation for Environmental Education and the RESOLVE Program of the World Wildlife Fund. 56 p.

Seim WK, Lichatowich JA, Ellis RH and Davis GE, 1977. Effects of Kraft mill effluents on juvenile salmon production in laboratory streams. Water Res. 11, 189-196.

Siefert RE, Lozano SJ, Brazner JC and Knuth ML, 1989. Littoral enclosures for aquatic field testing of pesticides: effects of chrorpyrifos on a natural system. In: Voshell JRJr (ed.), Using mesocosms to assess the aquatic ecelogical risk of pesticides: theory and practice. Entomol. Society of America, Misc. Publ. No. 75.

Sigmon CF, Kania HJ and Beyers RJ, 1977. Reductions in biomass and diversity resulting from exposure to mercury in artificial streams. J. Fish. Res. Board Can. 34, 493-500.

Stay FS, Katko A, Rohm CM, Fix MA and Larsen DP, 1989. The effects of atrazine on microcosms developed from four natural plankton communities. Arch. Environ. Contam. Toxicol. 18, 866-875.

Steber J, Guhl W, Stelter N and Schröder FR, 1995. Alkyl polyglycosides - ecological evaluation of a new generation of nonionic surfactants. Tenside 32, 515-522.

Steinman AD, Mulholland PJ, Palumbo AV, Deangelis DL and Flum TL, 1992. Lotic ecosystem response to a chlorine disturbance. Ecological Applications 2, 341-355.

Stephenson GL, Kaushik NK, Solomon KR and Day K, 1986. Impact of methoxychlor on freshwater communities of plankton in limnocorrals. - Environ. Toxicol. Chemistry 5, 587-603.

Stephenson RR, Mitchell GC, Pearson N, Worden J and Parker JJ, 1992. Development and validation of methods for evaluating chronic toxicity to freshwater ecosystems. CEC Contract No. EV4V-0110-UK(BA). Final Report (Summary), Commission of European Communities, Brussels, Belgium.

Stout RJ and Cooper WE, 1983. Effect of p-cresol on leaf decomposition and invertebrate colonization in experimental outdoor streams. Can. J. Fish. Aquat. Sci. 40, 1647-1657.

Stout J and Kilham SS, 1983. Effects of p-cresol on photosynthetic and respiration rates of a filamentous green alga (*Spirogyra*). Bull. Environ. Contam. Toxicol. 30, 1-5.

Strickland JDH and Terhune LDB, 1961. The study of *in-situ* marine photosynthesis using a large plastic bag. Limnol. Oceanogr. 6, 93-96.

Tagatz ME and Ivey JM, 1981. Effects of fenvalerate on field- and laboratory-developed estuarine benthic communities. Bull. Environm. Contam. Toxicol. 27, 256-267.

Tagatz ME, Ivey JM, Gregory NR and Oglesby JL, 1981. Effects of pentachlorophenol on field- and laboratory-developed estuarine benthic communities. Bull. Environm. Contam. Toxicol. 26, 137-143.

Tagatz ME, Deans CM, Moore JC and Plaia GR, 1983. Alterations in composition of field- and laboratory-developed estuarine benthic communities exposed to di-n-butyl phthalate. Aquat. Toxicol. 3, 239-248.

Tattersfield LJ, Holt M and Girling AG, 1995. The fate and effects of linear alkylbenzene sulphonate (LAS) in outdoor artificial streams and pools. Shell Research Report SBER,95 9.

Taub FB, Kindig AC and Conquest LL, 1986. Preliminary results of interlaboratory testing of a standardised aquatic microcosm. In: Cairns JJr (ed.), Community toxicity testing, STP 920, ASTM.

Tomlin C (ed.), 1994. The pesticide manual. Tenth edition. British Crop Protection Council and The Royal Society of Chemistry, Surrey and Cambridge, UK.

Topping G, 1977. Measurement and effects of sub-lethal levels of some heavy metals on marine life in simulated marine ecosystems. Proc. Analyt. Div. Chem. Soc., Aug. 1977, 222-224.

Uhlmann D, 1985. Scaling of microcosms and the dimensional analysis of lakes. Int. Rev. der gesamten Hydrobiologie 70, 47-62.

Unilever, 1989. The effects of LAS on the early life stages of rainbow trout. Unpublished report No. CT/MAR/07/RT, Unilever Research, Wirral, UK.

US-EPA (1984). Estimating "concern levels" for concentrations of chemical substances in the environment. Environmental Health Effects Branch, United States Environmental Protection Agency, Washington, DC.

Van de Plassche EJ, Belanger SE, Marshall SJ and Stephenson RR, 1997. Derivation of predicted no-effect concentrations for the surfactants lineat alkyl benzene sulphonate, alcohol ethoxylates, alcohol ethoxysulphonates and soap. In press.

Vailati G, Calamari D and Marchetti R, 1975. Effect of alkylbenzene sulfonate (LAS) on the development stages of *Salmo gairdneri.* Nuovi Ann. Ig. Microbiol. 26, 69-84.

Vargo GA, Hutchins M and Almquist G, 1982. The effect of low, chronic levels of No.2 fuel oil on natural phytoplankton assemblages in microcosms: 1. Species composition and seasonal succession. Marine Environ. Res. 6, 245-264.

Vial A, 1990. Report on the reproduction test of G 24480 technical to Daphnia (*Daphnia magna* Straus 1820). Internal report No. 891520. Ciba-Geigy Ltd., Basle.

Warren CE, Wales JH, Davis GE and Douderoff P, 1964. Trout production in an experimental stream enriched with sucrose. J. Wildlife Management 28, 617-660.

Watton AJ and Hawkes HA, 1984. Studies on the effects of sewage effluent on gastropod populations in experimental streams. Water Res. 18, 1235-1247.

Weber CI and McFarland BH, 1981. Effects of copper on the periphyton of a small calcareous stream. In: Bates JM and Weber CI (eds.), Ecological Assessments of Effluent Impacts on Communities of Indigenous Aquatic Organisms. ASTM STP 730, American Society for Testing and Materials, Philadelphia, Pennsylvania, pp 101-131.

Wilde EW and Tilly LJ, 1981. Structural characteristics of algal communities in thermally altered artificial streams. Hydrobiologia 76, 57-63.

Williams LG and Mount DI, 1965. Influence of zinc on periphytic communities. Amer. J. Bot. 52, 26-34.

Woltering DM and Bishop WE, 1989. Evaluating the environmental safety of detergent chemicals: a case study of cationic surfactants. In: Paustenbach DJ (ed.), The Risk Assessment of Environmental and Human Health Hazards. John Wiley & Sons, pp. 345-389.

Wrenn WB and Grannemann KL, 1980. Effects of temperature on bluegill reproduction and young-of-the-year standing stocks in experimental ecosystems. In: Giesy JP (ed.), Microcosms in Ecological Research 52, U.S. Department of Energy, 703-723.

Wrenn WB, Armitage BJ, Rodgers EB and Forsythe TD, 1979. Browns Ferry biothermal research series. II. Effects of temperature on bluegill and walleye, and periphyton, macroinvertebrate, and zooplankton communities in experimental ecosystems. EPA-600/3-79-092. Ecological Research Series Report, US Environmental Protection Agency, Athens, Georgia.

Yasuno M, Sugaya Y and Iwakuma T, 1985. Effects of insecticides on the benthic community in a model stream. Environmental Pollution Series A, 38, 31-43.

Yasuno M, Asaka A, Kono Y, 1993. Effects of pyraclofos (an organophosphorus insecticide) on nutrient enriched ecosystems. Chemosphere 27, 1813-1824.

Zischke JA and Arthur JW, 1987. Effects of elevated ammonia levels on the fingernail clam, *Musculium transversum*, in outdoor experimental streams. Arch. Environ. Contam. Toxicol. 16, 225-231.

Zischke JA, Arthur JW, Nordlie KJ, Hermanutz RO, Standen DA and Henry TP, 1983. Acidification effects on macroinvertebrates and fathead minnows (*Pimephales promelas*) in outdoor experimental channels. Water Res. 17, 47-63.

Zischke JA, Arthur JW, Hermanutz RO, Hedtke SF and Helgen JC, 1985. Effects of pentachlorophenol on invertebrates and fish in outdoor experimental channels. Aquatic Toxicol. 7, 37-58.

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