An Approach for Assessing the Water Quality Significance of Chemical Contaminants in Urban Lakes*

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Introduction

Increasing attention is being given to management of water quality in urban lakes. While these waterbodies have previously been generally neglected by water pollution control agencies, the US EPA and state and local governmental agencies are beginning to provide funding to investigate sources of contaminants for urban lakes and to develop water quality control programs for them. Two of the major forces behind these efforts are the US EPA Urban Lakes and the Combined Sewer Overflow Programs. These programs, if carried out properly, will provide municipalities with the information needed to develop cost-effective, technically valid water quality management programs for urban lakes. This paper considers a number of factors that should be evaluated in the development of such programs.

Characteristics of Urban Lakes

The first issue that must be addressed in the development of urban lakes' water quality management strategy is whether or not urban lakes behave differently from rural lakes with respect to contaminant load-water quality response. Generally urban lakes tend to be shallow, with maximum depths on the order of a couple of meters. With few exceptions they are highly fertile and would generally be classified as eutrophic. The littoral areas of many urban lakes are dominated by macrophytes (water weeds) and attached algae. They normally have "pea soup" green to brown color arising from planktonic algae and suspended sediment. Occasionally they will have dense algal blooms which can create obnoxious surface scum and odors due to decay of algae. Thermal stratification, if it develops, is usually short-lived; rarely would a thermocline persist throughout the summer growing season. With few exceptions, urban lakes have appreciable populations of rough fish such as carp and bullheads. All of these characteristics tend to make urban lakes fall into the group of shallow, weedy, moderately-to-highly turbid water-bodies which have appreciable mixing between the sediments and overlying waters. The overall water quality in urban lakes would normally be classified as poor. However, many urban lakes have considerable fishing and other recreational pressures placed on them. It is imperative that an understanding of the factors controlling urban lake water quality be achieved in order to maximize water quality improvements for funds spent in control programs.

In addition to water quality problems there is also the aesthetic problem of littering of the shores and bottom of urban lakes by the public. Unless effectively policed, the littering can be a significant deterrent-to the public's use of urban lakes.

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In those municipalities where salt is used for road deicing, the first major snow melt in the spring can bring appreciable amounts of salt into the lake. Some urban lakes have become meromictic (temporarily or permanently stratified) because of the accumulation of salt in the bottom waters of the lake.

While most urban lakes are freshwater, in coastal areas some are marine-estuarine which may be subject to the influence of the tide. In some municipalities, urban lakes are dammed rivers and therefore may have appreciable contaminant input from upstream sources. Many urban lakes are used as receptacles for combined sewer overflow in municipalities where combined sewers exist. Routinely they are the receptacle for urban stormwater drainage and frequently they receive domestic wastewater inputs from separated sewerage systems during periods of high flow, electrical outages, pump failure, etc.

Water Quality Management in Urban Lakes

The first step in the development of a water quality management program for any waterbody is an assessment of the water quality problems that exist within the waterbody. Generally, the most significant and in many cases the only water quality problem that exists in urban lakes is excessive fertilization and associated water weed and algae growth. Many urban lakes may receive relatively large loadings of a variety of potentially toxic chemicals such as heavy metals, pesticides, etc., these contaminants may cause water quality problems in urban lakes. However, more eutrophic waterbodies with high suspended sediments are able to "detoxify" to some extent many contaminants as a result of contaminant interaction with suspended and deposited particulate matter and aquatic vegetation.

Urban lakes tend to receive relatively large loads of oxygen demanding materials (BOD) in urban stormwater drainage and combined sewer overflows and/or domestic wastewater bypasses to the lakes. While oxygen demand problems can occur in urban lakes, they tend to occur just after a stormwater runoff event. The overall trophic status of many of these waterbodies is such that rarely would low dissolved oxygen be a significant cause of water quality deterioration during ice-free periods. However, as discussed by Lee and Jones (1991), in the winter in areas where waterbodies develop an ice cover, shallow lakes of this type, especially during periods of snow cover preventing penetration of light through the ice, tend to be prone to “winter kill” where more desirable game fish would be lost because of low dissolved oxygen.

Urban stormwater drainage and the entrance of domestic wastewaters into urban lakes tends to cause elevated fecal coliform counts which can serve as a basis for closing beaches and restricting other body contact with the water. Urban lakes in regions where there is appreciable new construction can receive large amount of erosional materials which will result in more rapid siltation especially near the point where the stormwaters enter the waterbody. These situations can cause water discoloration due to the presence of large amounts of suspended sediment in the urban stormwater drainage.

While it is possible to generalize on water quality problems of urban lakes, the water quality problems of any particular urban lake must be assessed on a case-by-case basis in which deterioration of water quality is judged based on impairment of a beneficial use of the water. Water quality should not be judged based on the concentration of a chemical contaminant in the water or the sediment. Many pollution control agencies make an error in assuming that one can relate water quality to the chemical composition of the water as normally determined by standard chemical analytical procedures normally used in the water quality field compared to water quality standards. Chemical contaminants exist in aquatic systems in a variety of chemical and physical forms, only some of which are available to affect water quality. Basing an urban lake water quality control program on the total contaminant load or concentration for a particular contaminant can result in the public spending large amounts of money in the name of water pollution control with little or no improvement in urban lake water quality. Those responsible for developing water quality management programs for urban lakes should first assess and document how the public's use of the urban lake is impaired by a particular water quality characteristic. Once this is known, it is then possible to begin to formulate the studies that are needed to provide the information which can serve as a basis for developing a water quality management strategy for a particular waterbody. These same issues apply to developing an urban lake in a new residential development.

Water Quality Criteria and Standards

The traditional approach that has been followed in the water quality control field used as a basis for developing water quality management programs has involved the use of worst case based water quality criteria and/or standards to evaluate water quality. Generally fixed numeric values have been established against which results of chemical analyses of water and/or sediments have been compared. This approach has worked reasonably well for the control of water quality deterioration arising from many municipal and industrial point source contaminants, where the objective of the control program was the elimination of gross water quality deterioration such as aquatic organism acute toxicity. Public Law 92-500, the 1972 Amendments to the Water Pollution Control Act, required that the US EPA develop national water quality criteria which are to become the backbone of the water pollution control program in the U.S. The US EPA released the first set of these criteria, commonly called the "Red Book" criteria, in July, 1976 (US EPA, 1976). These criteria have been updated several times with the most recent version being available on the US EPA website, http://www.epa.gov/waterscience/criteria/. Lee and Jones-Lee (1995) discussed how these criteria should be used to regulate sources of pollutants that impair the water quality of waterbodies.

These criteria originally and today are "worst case" values developed by exposing organisms for chronic - usually life-time periods of time to forms of the contaminants which are 100% available. They are designed to protect fish and other aquatic life against some of the more subtle effects of contaminants such as impairment of reproduction and growth rate. For many chemicals there is a several order of magnitude difference between the critical concentration which will kill fish within 96 hours and the chronic safe concentration. Adoption of the US EPA July, 1976 Red Book and subsequently developed criteria as water quality standards will, in general, provide the basis for ecologically protective water quality management programs. It is likely that there will be few instances where a waterbody containing concentrations of contaminants equal to or less than these criteria numeric concentrations would experience deteriorated water quality due to chemicals.

While ecologically protective, there are issues on the cost-effectiveness of application of standards numerically equal to the worst case criteria since that approach does not consider the fact that substantial parts of many chemical contaminants present in aquatic systems are in chemical forms which are unavailable to affect aquatic organisms and water quality. This is especially true for contaminants associated with particulate matter. Lee et al. (1978a), Jones and Lee (1978b), and Jones et al. (1979) have conducted a comprehensive study of the water quality significance of chemical contaminants associated with aquatic sediments from locations across the U.S. It has been found that most of the contaminants associated with aquatic sediments are in forms unavailable to affect water quality. They have pointed out that there is no relationship between the total concentration of a contaminant in a sediment and the potential for this contaminant to affect water quality. They further pointed out that it is not appropriate to use water quality standards numerically equal to US EPA criteria for many aquatic systems, especially those which contain large amounts of sediment-associated contaminants. Since the behavior of contaminants in urban lakes is frequently greatly influenced by sediments both suspended and deposited, application of water quality standards equal to US EPA water quality criteria would not in most instances be a cost-effective approach for developing a water quality management strategy for these systems.

Since the primary sources of contaminants for urban lakes are urban stormwater drainage and in some instances combined sewer overflow and domestic wastewater diversions, and since numerous studies have found that appreciable parts of the contaminants in these sources are in particulate forms, it is likely that water quality programs directed toward the total contaminant load to an urban lake could readily result in expenditure of large amounts of money for contaminant control with little or no improvement in urban lake water quality. Because of the relatively high cost associated with the control of contaminants in urban stormwater drainage, it is imperative that all funds spent for this purpose be directed to the maximum extent possible to the control of available forms of contaminants.

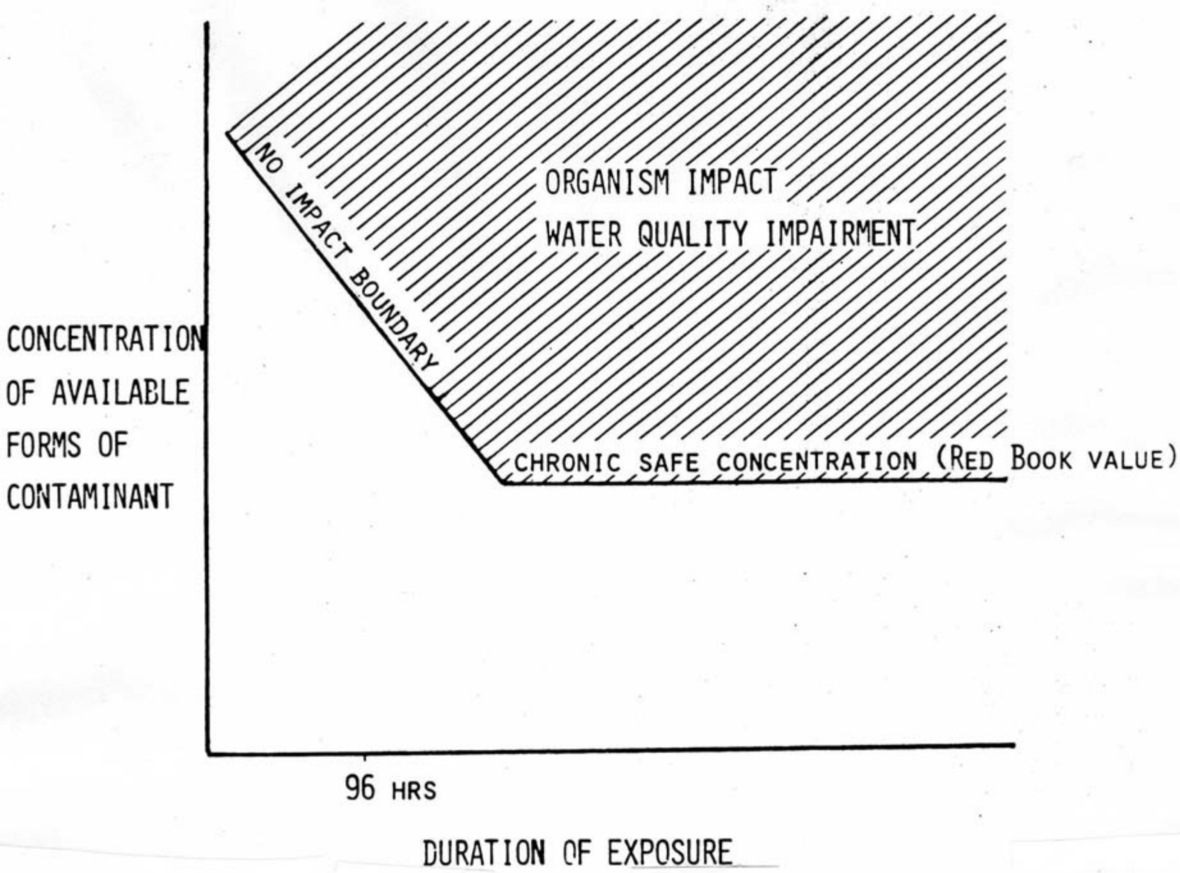
Hazard Assessment Approach

(*This discussion is adapted from Lee and Jones (1979b*)

In situations where chemical water quality standards numerically equivalent to the US EPA water quality criteria are not suitable indicators of water quality deterioration, such as in assessing the water quality impact of contaminants in urban stormwater drainage on urban lakes, it is necessary to use the hazard assessment approach to develop a technically valid, cost-effective, ecologically protective program for water quality management. This approach was originally developed for prescreening new chemicals for their environmental impact. Several papers in the book edited by Cairns et al. (1978) described the application of this approach for this purpose. Lee and Jones (1979a,b) and Lee et al. (1979b) have described the application of this approach for existing chemicals. Pollution control agencies will have to follow an approach similar to that discussed by Lee and Jones (1979a,b) and Lee et al. (1979b) in developing water quality control programs for urban lakes if these programs are to maximize water quality improvement for the money spent.

An aquatic environmental hazard assessment is built on two basic components: aquatic toxicology and environmental chemistry fate information. In the aquatic toxicology phase, the concentrations of the contaminant, effluent, or other source (such as urban stormwater drainage) of concern which cause no adverse impact on aquatic organisms is established. As part of this evaluation, concentration-duration of exposure couplings are developed for each contaminant or potentially hazardous input which define the "no adverse effect" level for a spectrum of exposure durations which may be encountered in the aquatic environment. Figure 1 shows a general-case concentration-duration of exposure relationship. It shows that for short durations of exposure, organisms can tolerate higher concentrations of contaminants without being adversely affected. As the exposure duration increases, the no effect concentration decreases eventually to a chronic safe level. For particular contaminants, these couplings must be based on concentrations of available forms; for a specific effluent containing a mixture of potentially hazardous materials, this can be based on dilution of effluent or other source of contaminants. In the aquatic toxicology phase of hazard assessment the bioaccumulation potential of the contaminant is also assessed. Assessment of this potential impact will be discussed further in a subsequent section.

Figure 1. General Available Concentration - Duration of Exposure Relationship



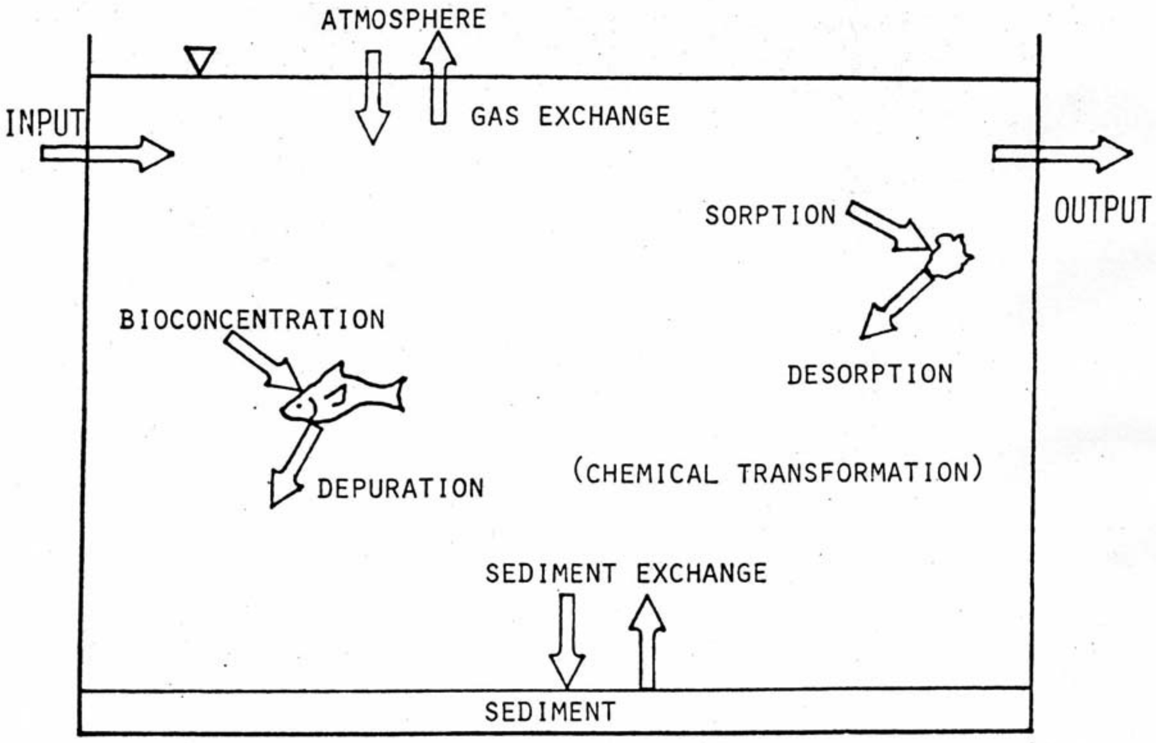
After Lee *et al*. (1979b).

The environmental chemistry-fate portion of an environmental hazard assessment considers the chemical processes that occur within the aqueous system as well as interactions with other environmental components that alter the form of the contaminant of concern and its transformation products in the aquatic system. Major types of chemical reactions that commonly occur in the aquatic environment which could cause significant changes in the form of a potentially hazardous chemical in terms of its water quality impact are acid-base, precipitation, complexation, oxidation-reduction, hydrolysis, photolysis, gas transfer, biochemically mediated reactions - biotransformation, and sorption, both biotic and abiotic. Jones-Lee and Lee (2008) have discussed the approach that should be followed to evaluate the water quality impacts of chemicals in urban stormwater runoff. The environmental chemistry-fate phase of a hazard assessment for an aquatic system also determines the transport pathways of the contaminant and its transformation products from the point at which it enters the aquatic system to its final disposition or the point at which it leaves the system. In association with this, the physical processes of advection-transport, and dilution-dispersion must be defined for the terrestrial, atmospheric, and aquatic environments with which the contaminant of concern and its transformation products come in contact. All of the pertinent environmental chemistry and fate information is formulated into a series of differential equations which describe each potentially significant transformation and transport pathway. Such an environmental chemistry-fate model is described schematically in Figure 2. Once an environmental chemistry-fate model has been verified it can be used to predict for a particular contaminant input, the concentration of the contaminant of concern and each potentially significant transformation product in each aquatic environmental component, i.e., in solution, associated with particulates, and associated with organisms, etc.

During the process of making an assessment of the environmental hazard associated with the particular source of contaminants, the predicted concentration-duration of exposure coupling is compared with the "no effect concentration-duration of exposure" relationship developed for the chemical or source in question in the toxicological portion of the hazard assessment (Figure 1), Figure 3 shows schematically a variety of the possible couplings and their relationship to the area of "impact." The hatched area in Figure 1 represents an area in which there is sufficient duration of exposure to sufficient concentrations of available forms of contaminant to have an adverse effect on aquatic organisms and/or water quality. This relationship must be defined for each contaminant or source of contaminants of concern in formulating a hazard assessment. The numbered curves (1-5) in Figure 3 show results that could be obtained through environmental chemistry-fate modeling, where a combination of dilution and chemical reactivity bring about a certain concentration-duration of exposure relationship. Curve 1 represents that coupling typical of spill situation, where there is toxicity for a short time associated with the point of entry before any reactions or dilution takes place. This might also be the situation associated with the mixing zone for a particular discharge such as urban stormwater drainage or combined sewer overflow.

Curve 2 in Figure 3 is a case where the contaminant at levels that are normally found in the environment does not show any acute toxicity but is chronically toxic either to an organism or to higher forms that may use the organism as food. PCB's, DDT, and mercury would all fall into this category. However, because of bioconcentration of some of these types of chemicals within the higher trophic level fish, there is a potential for harm to man and other animals that use these fish as a source of food.

Figure 2. Environmental Chemistry - Fate Model



$$\frac{D(\text{AVAIL. FORM})}{DT} = K_1(\text{GAS EXCHANGE}) + K_2(\text{BIOCONCENTRATION}) + K_3(\text{SORPTION}) + K_4(\text{CHEMICAL TRANSFORMATIONS}) + K_5(\text{ETC.})\ldots$$

After Lee _et al_. (1979b).

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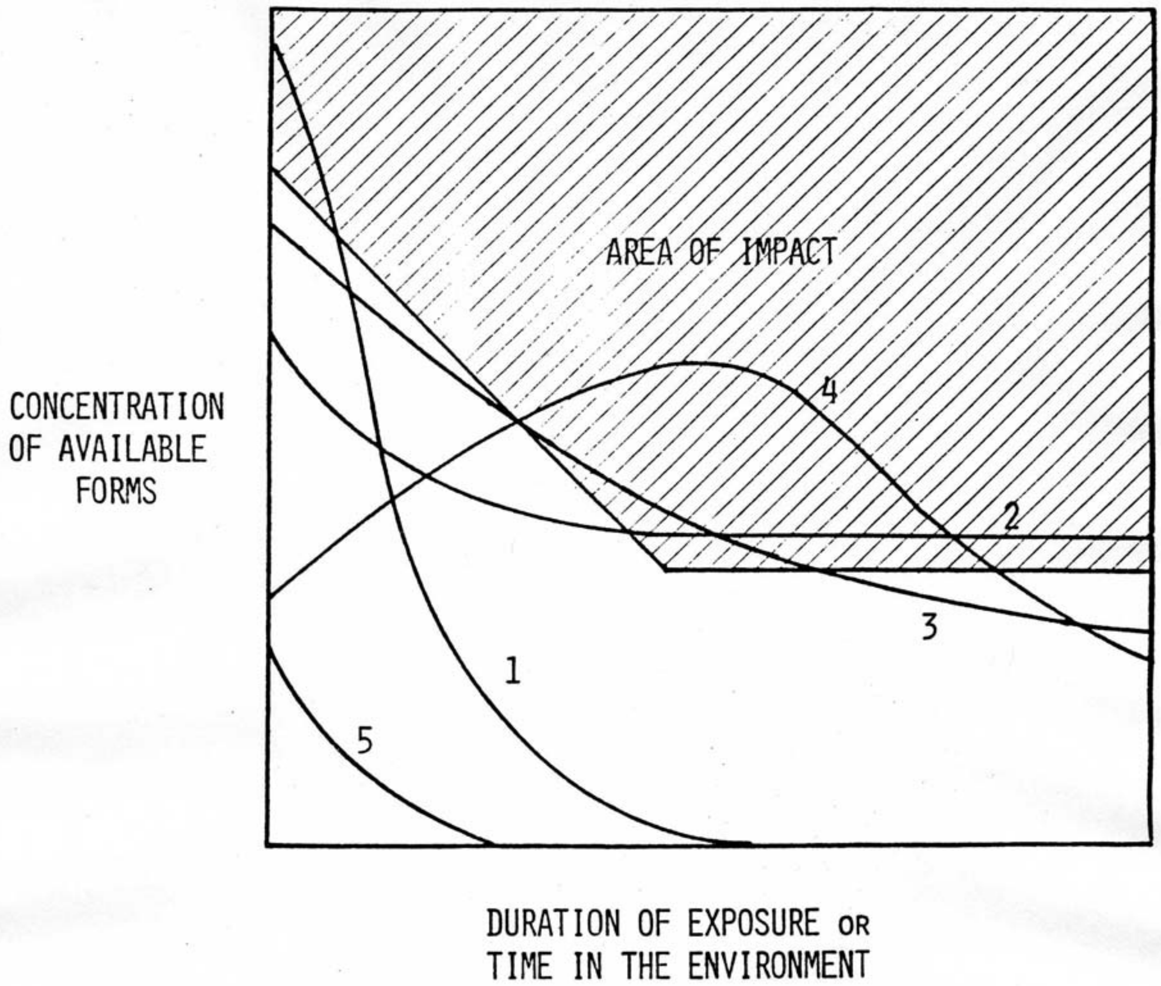
Figure 3. Concentration of Available Forms - Duration of Exposure Couplings



After Lee _et al_. (1979b).

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Curve 3 in Figure 3 represents the type of situation which might be associated with municipal wastewater discharges which contain ammonia, where for short durations of exposure there is no impact because possible durations of exposure are too short. However, there is an intermediate zone at some distance from the point of discharge where expected durations of exposure of the organisms to available forms are sufficient, where there could be toxicity to fish or other organisms that reside in the area. Eventually, the ammonia would be oxidized or diluted to non-toxic levels as one proceeds further down the stream from this zone.

Curve 4 is representative of the situation where there is a transformation of the contaminant added to the system which causes it to be more toxic as it goes downstream. Eventually it is either diluted or detoxified through other reactions. An example of this type of situation is one involving the addition of a complexed heavy metal to the environment where the complex is biodegradable, releasing the heavy metal at some distance downstream in sufficient concentrations to be toxic to aquatic life in that region. Curve 5 is the case that exists for most chemicals for which there is sufficient treatment or controlled use so that there is no toxicity associated with it, either acute or chronic.

An environmental hazard assessment for any contaminant source should be conducted in a series of levels or tiers in which the aquatic toxicology and environmental chemistry-fate are determined with increasing sophistication and reliability with succeeding tiers, with a decision point at the end of each tier. The decision choices are: 1) do not allow discharge because of excessive expected hazard, 2) restrict discharge through use or treatment to reduce environmental hazard to an acceptable level, 3) proceed with discharge as is - expected environmental hazard is acceptable, or 4) continue testing to more precisely defined expected impact. The overall approach is to screen for gross effects of a contaminant such as acute toxicity and estimated chronic toxicity in the earlier tiers. At the higher tiers, the focus is on contain ammonia, where for short durations of exposure there is no impact because possible durations of exposure are too short. However, there is an intermediate zone at some distance from the point of discharge where expected durations of exposure of the organisms to available forms are sufficient, where there could be toxicity to fish or other organisms that reside in the area. Eventually the ammonia would be oxidized or diluted to non-toxic levels as one proceeds further down the stream from this zone.

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toxicity and estimated chronic toxicity in the earlier tiers. At the higher tiers, the focus is on conducting tests for the more subtle effects of the contaminant such as impairment of reproduction. These higher tier tests only need be conducted if the lower tiers suggest that there is a potential for this type of water quality problem. As shown in Figure 4, with each succeeding tier of testing a more precise estimate of both the actual expected environmental concentration and also the "no impact" concentration for organisms is obtained.

The tiered environmental hazard assessment approach will likely be of particular significance in developing water quality management plans for urban lakes. It is likely that the results of a few screening bioassays normally associated with lower tiers of testing will show that urban stormwater drainage does not contain sufficient amounts of contaminants to cause toxicity. Findings of this type can greatly simplify the study program and thereby reduce its cost compared to the normal procedure for studying water quality involving routinely collecting data for a year or two and then attempting to interpret the data at the end of the study period in terms of the waterbodies water quality. Most water quality studies as conducted today provide limited amounts of information that can be used to formulate cost-effective, technically valid and at the same ecologically protective water quality management programs.

Control of Toxics

A key component of the hazard assessment approach for urban lakes as well as other waterbodies is the determination of the availability of the contaminants in the inputs to the lake. As discussed by Lee et al. (1979a), Jones and Lee (1978b), and Jones et al. (1979), chemical leaching tests for particulate matter such as are frequently used, do not in general properly assess the amounts of available forms of contaminants that can affect water quality for conditions typically found in urban lakes. Instead, bioassay procedures must be used on the sources of contaminants because of the variety of contaminants in the sources and for urban lakes, the typically high suspended particulate levels. If chemical leaching tests are used, the results must be verified against an extensive series of bioassays conducted at various times of the year in order to determine if there are any seasonal effects on the availability of the contaminants.

The first tier of an environmental hazard assessment for toxicants associated with contaminant inputs to urban lakes should be a 96 hour static bioassay screening test of the type described by Lee et al. (1978a) on the undiluted water from each input. If no acute toxicity is found in these waters, and if there is a population of warm water pan fish (i.e., bluegill, sunfish, etc.) which are reproducing in the waterbody, and if there is at least a 100-fold dilution of the input waters with the lake water within a few days to a week or so of input, then the likelihood of there being any type of toxicity, including impairment of reproduction due to this source, is remote. The 100-fold dilution is based on the factor that is typically found to relate acute lethal toxicity to the chronic safe limit.

If acute toxicity is found in the screening bioassays of the undiluted urban stormwater drainage or other source of contaminant for an urban lake, then additional bioassays must be conducted on diluted contaminant source water, using appropriate dilutions with lake water based on the characteristics of the waterbody, to check for both acute and chronic toxicity. If potentially significant toxicity is found, it would be important to attempt to identify the specific toxic component. This can possibly be accomplished using the standard additions approach described by Lee and Jones (1979d). For further information on the use of bioassay procedures in a hazard assessment, consult Lee et al. (1979b).

Since bioaccumulation of contaminants in fish is of concern in urban lakes where the urban lake fish are used as food, it is imperative that some work also be done in the early tiers of any hazard assessment program to determine whether excessive buildup of contaminants is occurring in organisms used for human food. In general, the US EPA fish tissue human health guidelines, http://www.epa.gov/waterscience/criteria/ are appropriate for evaluating excessive bioaccumulation of a hazardous chemicals in edible aquatic life. In California, the updated OEHHA fish tissue guidelines, http://oehha.ca.gov/fish/so_cal/index.html are appropriate for screening for excessive bioaccumulation of hazardous chemicals in edible aquatic life. The most appropriate and only reliable method for assessing potential water quality problems associated with certain contaminants is to collect from the lake of concern seasonally over at least one year, representative samples of fish being used as human food and analyze their flesh for each of those contaminants for which such as PCB's, mercury, DDT and its analogs, as well as several other chlorinated hydrocarbon legacy pesticides are measured in edible organism tissue. If the concentrations of any of those contaminants would cause the fish to be unsuitable for use as food, then each source of that contaminant must be investigated and control programs initiated where possible.

It is important to note that for developing control programs, while first priority must be given to analyzing fish flesh for those chemicals for which human health consumption guidelines exist, since those are the only interpretable data at this time, as funds permit, the fish should be analyzed for any other contaminant which may be present in large amounts or may be of concern for a particular system. While these data may not at this time be interpretable in terms of water quality and fish usability, they may be a clue to potential water quality problems that should be investigated further since organisms in aquatic systems tend to be integrators for some types of persistent contaminants.

Control of Eutrophication

Nutrients (nitrogen and phosphorus) frequently cause significant water quality deterioration in the urban lake waters. The hazard assessment approach for evaluating the significance of nutrient inputs to waterbodies, including urban lakes, can be based on the OECD eutrophication modeling approach. During the 1970s, Organization for Economic Cooperation and Development (OECD) sponsored a 22 country, 200 waterbody (lake and impoundment) study of the relationships between the nutrient load to a waterbody and the eutrophication response of that waterbody. Based on concepts originally developed by Vollenweider (1968) and the data on the approximately 40 OECD waterbodies in the U.S., Rast and Lee (1978) found that good correlations existed between the P load to a waterbody, normalized by the waterbodies mean depth and hydraulic residence time, and the average summer planktonic algal chlorophyll concentrations, planktonic algal-related water clarity as measured by average summer Secchi depth, and for those waterbodies which thermally stratify, hypolimnetic oxygen depletion rate. Figure 5 shows these relationships. Included in this data set were several urban lakes including

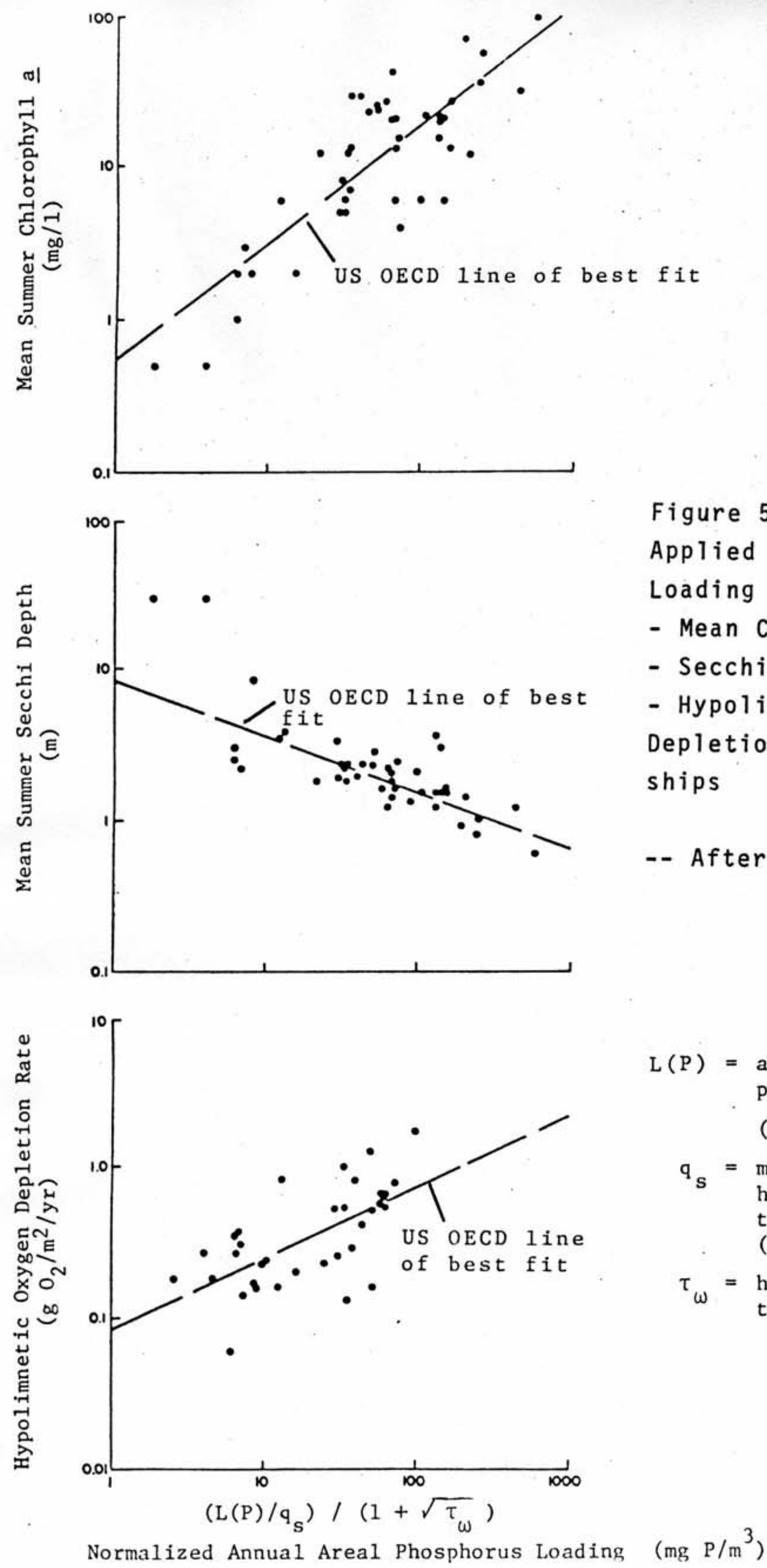

Figure 5. US OECD Data Applied to Phosphorus Loading
- Mean Chlorophyll a
- Secchi Depth, and
- Hypolimnetic Oxygen Depletion Rate Relationships

-- After Lee et al. (1978b)

KEY

$L(P)$ = areal annual phosphorus load (mg P/m^2/yr)

q_s = mean depth ÷ hydraulic residence time (m/yr)

τ_ω = hydraulic residence time (yr)

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Lake Wingra in Madison, Wisconsin. While some have claimed that the nutrient behavior in small shallow lakes of this type is not the same as that in larger, deep waterbodies, the OECD eutrophication model has demonstrated that when normalized by hydrologic and morphologic characteristics, small and large lakes and impoundments behave in a similar manner in terms of eutrophication response to nutrient inputs. Lee et al. (1977, 1978b) have presented summaries of these results for the US OECD waterbodies. Additional information on these issues is available on the Lee and Jones-Lee website, www.gfredlee.com in the Excessive fertilization section at, http://www.gfredlee.com/pexfert2.htm.

Subsequent to the completion of the Rast and Lee (1978) work, Lee and his associates have evaluated the nutrient load-response relationships for an additional 700 or so waterbodies located in many areas of the world and have found that these waterbodies follow the same relationships shown in Figure 5. Further, the other approximately 150 OECD waterbodies' nutrient load-eutrophication response relationships also are in agreement with what Rast and Lee (1978) found for the US OECD waterbodies.

Lee and Jones (1991), who have conducted an extensive review of the effect of eutrophication on fisheries, have used the data provided by Oglesby (1977) and Rast and Lee (1978) to develop the relationship between normalized P load and fish yield shown in Figure 6. As would be expected, it shows that the greater the nutrient load the greater the over-all fish yield. Lee and Jones (1991) should be consulted for further information on this topic.

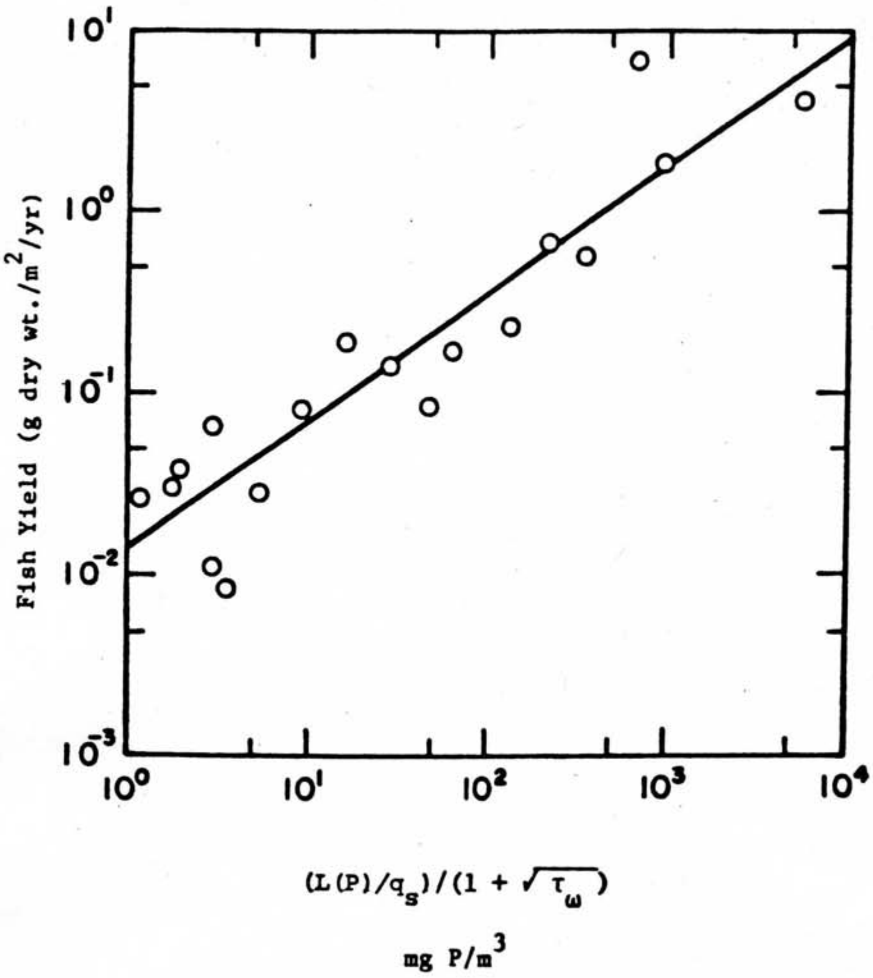
The OECD eutrophication modeling approach is also only applicable to waterbodies in which aquatic plant growth is dominated by phytoplankton. In its current degree of development it cannot be used to assess eutrophication response as measured by macrophytes or attached algal growth parameter. As discussed by Newbry et al. (1979), it is likely possible to develop nutrient load-response relationships for macrophytes where response is assessed in terms of the percent of the area of the waterbody with depth less than 2m that is covered by aquatic macrophytes and attached algae.

A constraint on the application of the OECD eutrophication model is that the average hydraulic residence time (i.e., filling time - volume divided by annual input) of the waterbody must be two weeks or more. For waterbodies having an annual hydraulic residence time shorter than two weeks, it may be possible to modify the model as was done by Jones and Lee (1978a) for Lake Lillinonah, CT. For this waterbody the summer average hydraulic residence time was used since each spring, the waterbody is essentially completely flushed.

Water Quality Management in Urban Lakes

The overall approach that should be used for management of water quality on urban lakes is the same as for any water-body, namely, limiting the input of available forms of contaminants to the extent necessary to achieve the desired water quality. In assessing available forms, it is important to assess not only those forms which are immediately available but also those which can become available in the waterbody. Because of the potential significance of urban stormwater drainage as a source of contaminants for urban lakes, in those situations where the hazard assessment shows that this source is a significant source of contaminants causing water quality deterioration, studies should be done to define the source of contaminants in the urban stormwater drainage. At this time there is a relatively poor understanding of the specific sources of total as well as available forms of contaminants in this source. The studies of Cowen and Lee (1973) and Kluesener and Lee (1974) have shown that in the fall, an appreciable part of the P present in urban stormwater removal of leaves and frequent street sweeping using vacuuming

Figure 6. Relationship between P load and Fish Yield



Line of best fit:

Fish Yield = 0.7 log $[(L(P)/q_s)/(1 + \sqrt{\tau_\omega})] - 1.86$

$(r^2 = 0.86)$

After Lee and Jones (1979c).

could significantly reduce the phosphorus content of urban stormwater drainage. Ahern and Armstrong (1979) found that street sweeping reduced the P content of urban stormwater drainage by 47 to 59%.

The importance of focusing nutrient control programs on available forms of nitrogen and phosphorus was demonstrated by the studies by Lee and his associates in Madison, WI. Kluesener and Lee (1974) found that urban stormwater drainage contributed about 0.1 g/m2/yr of total phosphorus and 0.5 g/m2/yr of total nitrogen. About 25% of the total N was in the form of ammonia and nitrate, while about 50% of the total P was in the form of soluble ortho P. Cowen et al. (1978), using bioassay techniques found that about 50% of the particulate organic and inorganic N (i.e., the difference between the total N and the sum of nitrate and ammonia) would likely become available to stimulate algal growth in an urban lake. They also found that only about 20% of the difference between the total P and soluble ortho P would likely become available to support algal growth. These results point to the fact that appreciable parts of the N and P entering urban lakes from urban stormwater drainage are in forms not likely to support algal growth in the waterbody. While the results of the studies by Cowen et al. (1978) are in general agreement with what other investigators who have conducted similar studies have found, additional work needs to be conducted for each system of interest until sufficient information is gathered to make appropriate estimates of nutrient export from urban areas and the availability of these nutrients to support algal growth. Lee et al. (1980) have presented a summary of information on available phosphorus measurement and typical amounts found in runoff.

A potentially significant source of nutrients for some urban lakes is waterfowl. While some of these birds only alter recycle of nutrients within the waterbody, such as ducks which feed on aquatic organisms, other waterfowl such as geese feed on land but spend most of their time in the water. The latter type of bird can add nutrients from on-land sources which may never otherwise reach the lake. An example of the potential importance of this problem is seen in urban lakes in the City of Fort Collins, Colorado, where the population of resident geese number well into the tens of thousands.

Typical urban stormwater and/or combined sewer contaminant control programs are usually directed toward the control of particulate forms of contaminants. Most of these programs are based on the erroneous premise that there is a relationship between the total load of a contaminant and its impact on water quality. As discussed above, it is rare that the total load or concentration of a contaminant can be used to predict the impact of the contaminant on water quality. While control of particulate forms of a contaminant may be useful for siltation control, it may not be cost-effective for reducing the magnitude of chemical related water quality problems. In many instances the most cost-effective way to deal with siltation, is periodic dredging.

The first priority in any urban lake's water quality maintenance program should be the removal of debris-trash usually left by the users of the area. The next priority should be given to the control of excessive fertility. Obviously, all significant point sources of nitrogen and/or phosphorus depending on what is limiting planktonic algal growth in the waterbody, should be identified. If the hazard assessment study shows P concentrations are in the range that could be limiting or can be reduced to growth rate limiting through point source control, and the point source(s) already has some treatment for removal of solids and BOD, then P removal by alum or iron co-precipitation techniques should be added to the treatment process. Generally, P can be controlled in domestic wastewater to less than 1 mg P/L total P at a cost of less than a half a cent (0.5 cent) per person per day for the population served. As noted above, the OECD eutrophication study modeling results can be used to estimate the magnitude of planktonic algal-related water quality improvement associated with point source and diffuse source nutrient control. Before embarking on large-scale street vacuuming or other diffuse source control programs, the magnitude of water quality improvement that could be achieved by these measures should be determined using the OECD eutrophication model. Such control programs are often relatively expensive and frequently provide limited reductions in available nutrient loads. As shown in Figure 5, relatively large reductions in P load are needed to significantly change the planktonic algal-related chlorophyll concentration. This is especially true for eutrophic lakes in which even relatively large reductions in chlorophyll may not result in a perceivable improvement in water quality.

As a result of their shallow character, many urban lakes have significant water quality problems due to attached algae and macrophytes. As discussed by Lee (1973) a variety of techniques such as aquatic herbicides, harvesting, etc., are available for control of these growths. In general, the overall control philosophy for aquatic macrophytes and attached algae should be to remove the fewest number of these plants as necessary to make the waterbody usable for the desired recreational purposes. Aquatic macrophytes and especially attached algae compete with planktonic algae for available nutrients. Extensive removal of attached algae and macrophytes can result in production of phytoplankton blooms and could destroy fish habitat and nursery grounds.

The dredging of urban lakes for the removal of deposited sediment can also have a significant impact on the relative utilization of nutrients by various types of aquatic plants. Extensive dredging to depths of 3m or more can greatly decrease the macrophyte and attached algae in eutrophic lakes with a concomitant increase in phytoplankton. Dredging should be restricted to those parts of the waterbody for which the shallowness permits excessive growths of attached algae and macrophytes which significantly interfere with recreational uses of the water such as swimming, boating, or fishing.

The direct alum addition to urban lakes is a technique that deserves much greater attention for managing urban lake water quality than it has received in the past. In many situations it may be the only technique available to manage excessive fertilization problems. Since urban lakes tend to normally be turbid, alum flocculation of inflowing waters can help reduce turbidity and also remove P from the water column. The alum floc that forms can be settled either directly in the lake or in a pre-impoundment constructed for this purpose. Systems can be developed to automatically feed alum to the stormwater, combined sewer overflow, or domestic wastewater bypass inputs to the lake. This approach does not appear to be harmful to fish as long as there is adequate alkalinity in the water to maintain a desirable pH. For low alkalinity waters, it may be necessary to add lime to increase the buffer capacity of the water.

The duration of effectiveness of this approach is directly dependent on the hydraulic residence time of the waterbody. Waterbodies with several years residence times will require relatively infrequent treatment. However, the typical urban lake would require the more or less continuous treatment of all inflowing waters to be effective. It should be noted that such treatment may result in increased amounts of attached algae and macrophyte growth because of the greater water clarity arising from alum treatment. This may necessitate additional harvesting or other control programs especially for macrophytes which can derive at least part of their nutrients from the sediments.

Those responsible for developing nutrient control programs for waterbodies are frequently concerned about the rate of recovery of the waterbody upon altering the nutrient load. The shallowness of many urban lakes has led some to believe that these waterbodies will not recover upon reduction of nutrient load because of the release of nutrients from the sediment. As discussed by Sonzogni et al. (1976), the rate of recovery of a waterbody upon reduction of P input depends on the P residence time of the waterbody. About 95% of the expected recovery from the degree of altered load occurs within a period equal to three times the phosphorus residence time. The P residence time of many waterbodies is less than one year which means that within a few years after altering the load, a new equilibrium P-chlorophyll concentration will be achieved for the waterbody.

Conclusions

Increasing pressures are being placed on water pollution control agencies to manage water quality in urban lakes. The focus of these programs is urban stormwater runoff management. The first step in developing a water quality management plan for such a waterbody is a definition of the water quality problems that may exist therein. In this evaluation, for many urban lakes, particular attention should be given to eutrophication-related water quality which will require reduction in the amounts of available forms of nitrogen and phosphorus entering them. The OECD eutrophication study results can be used to describe the nutrient load-response relationships for the urban lakes and most importantly, should be used to estimate the magnitude of water quality improvement that will arise as a result of altering the nutrient loads to P limited urban lakes. The issue of assessing limiting nutrients is discussed in the supplemental section of this paper.

In many instances it may not be possible to control nitrogen and/or phosphorus to a sufficient degree to attain desired water quality in urban lakes because of the difficulty of limiting nutrient input from urban stormwater drainage and/or combined sewer overflow. Direct alum addition to the inflowing waters is a technique that should be more thoroughly investigated as a tool to manage excessive fertility in urban lakes. Also selective dredging and attached algae and macrophyte control through harvesting can be effective tools for minimizing impact of aquatic plant nutrients on urban lake water quality.

For those waterbodies impacted by toxicants as well as nutrients, an environmental hazard assessment should be conducted to define the magnitude of the impact of contaminant sources on water quality, i.e., beneficial uses, of the lake. Such an assessment can also be used to develop the information necessary to control the excessive input of the contaminants. Water quality standards adapted directly from the US EPA water quality criteria and state standards have limited applicability in serving as a basis for developing water quality management programs for urban lakes. The environmental hazard assessment involving the tiered development of environmental chemistry-fate and aquatic toxicology information with emphasis given to use of bioassays to define available forms of the contaminant should be used.

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Supplemental Information
This paper was original developed in 1979. During the past 20 years Drs. G. Fred Lee and Anne Jones-Lee have developed considerable additional information that further refines the approaches that need to be considered in development of urban lakes. Their website,www.gfredlee.com in the Surface Water, and Excessive Fertilization sections contain their writings on these issues.

Limiting Nutrient Issues
In the previous presentations of the constraints in the use of the Vollenweider OECD modeling approach is that it is restricted to waterbodies with the phytoplankton biomass that are limited to those that are limited by available phosphorus concentrations. In previous writings the limiting nutrient in a waterbody or an area of a waterbody can be determined by measuring the concentrations of available N (NO_3^- plus NH_3) and available P (soluble ortho P) during the period of maximum phytoplankton biomass. In general, if the available P concentration is reduced to a few µg P/1, the phytoplankton growth at the time the samples were collected was most likely limited by P. If the available N concentrations are reduced to about 30 to 50 µg/1 or so, N is likely limiting phytoplankton production. If neither nutrient is reduced to these levels, some other factor is likely limiting maximum plank-tonic algal biomass.

The ratio of available N to available P is also used to indicate which nutrient would be depleted first (i.e., potentially limiting) in a water, based on the theoretical uptake ratio of these nutrients by algae of 7.5 N to 1 P on a mass/L basis. Algal assays are also used to estimate the limiting nutrient by determining which nutrient if added would promote increased algal growth. Caution must be exercised in using the latter two approaches to determine the limiting nutrient; they must be performed near the time of maximum algal production since the results of such tests run at other times of the year will not necessarily give an accurate representation of the limiting nutrient during the period of water quality concern. Further, analyses for available N and P during peak biomass production should be conducted in conjunction with these procedures to verify that one of these nutrients is actually limiting the growth; while an N-to-P ratio may indicate a lesser relative abundance of one nutrient, some other factor such as light may in reality limit algal growth during the period of concern.

Today it is now understood that available phosphorus considerable above the limiting concentrations developed using the approaches discussed above. Lee and Jones-Lee have developed,

Lee, G. F. and Jones-Lee, A., "Developing Nutrient Criteria/TMDLs to Manage Excessive Fertilization of Waterbodies," Proceedings Water Environment Federation, TMDL 2002 Conference, Phoenix, AZ, November (2002). http://www.gfredlee.com/Nutrients/WEFN-Criteria.pdf

which discusses the finding that the phytoplankton biomass in waterbodies responds to reduced phosphorus concentrations will above the phosphorus limiting concentrations based on N to P ratios at peak biomass.

Lee and Jones-Lee have further discussed this issue in,

Lee, G. F., and Jones-Lee, A., “Synopsis of CWEMF Delta Nutrient Water Quality Modeling Workshop – March 25, 2008, Sacramento, CA,” Report of G. Fred Lee & Associates, El Macero, CA, May 15 (2008). http://www.gfredlee.com/SJR-Delta/CWEMF_WS_synopsis.pdf

Presented below are excerpts from Drs Lee and Jones-Lee CWEMF presentation on limiting nutrient issues.

Excerpts from Impact of Altering the Phosphorus Loads to the Delta on Delta Algal Concentrations

At the CWEMF Delta Nutrient Water Quality Modeling Workshop, Dr. Erwin Van Nieuwenhuyse, Fishery Biologist with the US Bureau of Reclamation Division of Environmental Affairs, Sacramento, CA (evannieuwenhuyse@mp.usbr.gov) made a presentation, “Impact of Sacramento River Input of Phosphate to the Delta on Algal Growth” that discussed the impact of altering phosphorus loads to the Delta phytoplankton biomass. His presentation is available at CWEMF website, http://cwemf.org/Calendar/index.htm.

Van Nieuwenhuyse presented data describing the impact of reducing the phosphorus concentrations in the Rhine River in Europe on phytoplankton chlorophyll. He reported that there was a significant decrease in planktonic algal chlorophyll there associated with decreased phosphorus concentrations. He also pointed out that those data suggest that phosphorus concentrations of about 400 ng/L appear to be an upper limit of the concentration range in which decreasing the P concentration effects a reduction in planktonic algal biomass.

At the CWEMF workshop, Dr. Lee mentioned that Rast et al. conducted a review of the literature to assess and quantify the impacts of altering the phosphorus loads to waterbodies (and hence, in-waterbody concentrations) on phytoplankton concentrations. The results of their investigations are presented in:

Rast, W., Jones, R. A. and Lee, G. F., “Predictive Capability of US OECD Phosphorus Loading-Eutrophication Response Models,” Journ. Water Pollut. Control Fed. 55:990-1003 (1983). http://www.members.aol.com/annelhome/PredictiveCapabilityOECD.pdf

Rast et al.’s findings also support the position that for waterbodies with apparently surplus phosphorus compared to the typically reported phosphorus half-saturation-constant of a few nanograms per liter, phosphorus reduction can be expected to result in reduction of planktonic algal chlorophyll levels, up to a limit of a few hundred nanograms P per liter.