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Module 2: Beneficial Uses and Problems Associated with Stormwater, in Three Parts

I. The Beneficial Uses of Stormwater in Urban Areas and the Need for Change in Urban Water Management

II. Stressor Categories and their Effects on Humans and Ecosystems

III. Water Quality Conditions in the Cahaba River and Likely Pollutant Sources

Contents

I. The Beneficial Uses of Stormwater in Urban Areas and the Need for Change in Urban Water Management

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Stormwater has classically been considered a nuisance, requiring rapid and complete drainage from areas of habitation. Unfortunately, this approach has caused severe alterations in the hydrological cycle in urban areas, with attendant changes in receiving water conditions and uses. This historical approach of "water as a common enemy" has radically affected how urban dwellers relate to water. For example, most residents are not willing to accept standing water near their homes for significant periods of time after rain has stopped. However, there are now many examples where landscape architects have very successfully integrated water in the urban landscape. In many cases, water has been used as a focal point in revitalizing downtown areas. Similarly, many arid areas are looking at stormwater as a potentially valuable resource, with stormwater being used for beneficial uses on-site, instead of being discharged as a waste. One of the earliest efforts investigating positive attributes of stormwater was a report prepared for the Storm and Combined Sewer Program of the U.S. Environmental Protection Agency by Hittman Associates in 1968. Only recently has additional literature appeared exploring beneficial uses of stormwater. This section discusses some of these progressive ideas.

Stormwater as an Aesthetic Element in Urban Areas

Dreiseitl (1998) states that "stormwater is a valuable resource and opportunity to provide an aesthetic experience for the city dweller while furthering environmental awareness and citizen interest and involvement." He found that water flow patterns observed in nature can be duplicated in the urban environment to provide healthy water systems of potentially great beauty. Without reducing safety, urban drainage elements can utilize waters refractive characteristics and natural flow patterns to create very pleasing urban areas. Successful stormwater management is best achieved by using several measures together. Small open drainage channels placed across streets have been constructed of cobbles. These collect and direct the runoff, plus slow automobile traffic and provide dividing lines for diverse urban landscaping elements. The use of rooftop retention and evaporation reduce peak flows. Infiltration and retention ponds can also be used to great advantage by providing a visible and enjoyable design element in urban landscapes.

Dreiseitl (1998) described the use of stormwater as an important component of the Potsdamer Platz in the center of Berlin (expected to be completed by the end of 1998). Roof runoff will be stored in large underground cisterns, with some filtered and used for toilet flushing and irrigation. The rest of the roof runoff will flow into a 1.4 ha (3.8 acre) concrete lined lake in the center of the project area. The small lake provides an important natural element in the center of this massive development and regulates the stormwater discharge rate to the receiving water (Landwehrkanal). The project is also characterized by numerous fountains, including some located in underground parking garages.

Göransson (1998) also describes the aesthetic use of stormwater in Swedish urban areas. The main emphasis for this study was to retain the stormwater in surface drainages instead of rapidly diverting the stormwater to underground conveyances. Small, sculpturally formed rainwater channels are used to convey roof runoff downspouts to the drainage system. Some of these channels are spiral in form and provide much visual interest in areas dominated by the typically harsh urban environment. Some of these spirals are also formed in infiltration areas and are barely noticeable during dry weather. During rains, increasing water depths extenuate the patterns. Glazed tile, small channels having perforated covers, and geometrically placed bricks with large gaps to provide water passage slightly below the surface help urban dwellers better appreciate the beauty of flowing water.

Tokyo has instituted major efforts to restore historical urban rivers that have been badly polluted, buried or have had all of their flows diverted. Fujita (1998) describes how Tokyo residents place great value on surface waterways: "waterfront areas provide urban citizens with comfort and joy as a place to observe nature and to enjoy the landscape." Unfortunately, the extensive urbanization that has taken place in Tokyo over the past several decades has resulted in severe stream degradation and disappearance of streams altogether. However, there has recently been a growing demand for the restoration of polluted urban watercourses in Tokyo. This has been accomplished in many areas by improved treatment of sanitary sewage, reductions in combined sewer overflows and by infiltration of stormwater.

The Meguro and Kitazawa streams have been recovered by adding sanitary wastewater (receiving secondary treatment, plus sand filtration and UV disinfection, with activated carbon filtration and ozone treatment to provide further odor control) to previously dry channels. The treated wastewater is being pumped 17 km from the treatment facilities to the upstream discharge location in Meguro Stream. The Nogawa Stream has been restored by adding springwater produced from stormwater infiltration. Increased firefly activity has been noted along the Nogawa Stream and the adjacent promenade, providing adequate justification for these projects to the local citizens.

The quality of the treated wastewater entering Meguro Stream (at 0.35 m^3/s) since 1995 is as follows: total BOD_5: 6 mg/L; carbonaceous BOD_5: 2 mg/L; suspended solids: 0.5 mg/L; and ammonia-nitrogen: 7 mg/L. The total coliform bacteria concentrations were initially high (5,000 MPN/100 mL), and UV disinfection was therefore later installed at the outlets of the treated wastewater to the stream. The receiving water biological uses (carp and crustaceans) require the following conditions: total BOD_5: <8 mg/L; a water depth of at least 10 cm, and a stream velocity of at least 0.1 m/s. The BOD_5 goals are being met and the Meguro Stream has a 20 cm depth and a velocity of about 0.3 m/s. When storm events occur, remote valves are operated to decrease the discharge of the treated wastewater into the stream. However, the physical habitat of the stream is currently severely degraded, being concrete lined. The local residents are appreciative of the small flow in the stream, and the Tokyo Metropolitan Government (TMG) plans to modify the stream walls to facilitate groundwater recharge of the stream, to create rapids and pools for fish, and to plant trees along its banks, to further enhance the value of the stream to the local population.

Kitazawa Stream is another example of a severely degraded urban stream in Tokyo that has undergone extensive modification. The stream watershed is 10.5 km^2 and has a population of about 150,000 people. The rapid urbanization in Tokyo since the 1950s has resulted in a severe decrease in groundwater infiltration during rains. This has caused decreased groundwater levels and decreased the associated natural recharge into urban streams. By the 1960s, there was almost no natural flow in Kitazawa Stream during dry weather. The only flows present in the stream was wastewater from homes. The stream was therefore of extremely poor quality, creating an unsafe and nuisance condition. In addition, the increased development caused frequent flooding. The TMG therefore diverted the stream into an underground culvert. The aboveground area was converted into a promenade with extensive plantings. Recently however, local residents have requested the addition of a steam along the promenade. A very small flow (0.02 m^3/s) of treated wastewater has been pumped from 11 km away to create this new stream (a “two-storied watercourse”). Figure 2-1 (Fujita 1998) shows the changes that Kitazawa Stream has undergone as the watershed has developed. This new steam, however small, has created a very important element in the lives of the residents of this heavily urbanized city. Special community organizations have been established to plan and manage the area.

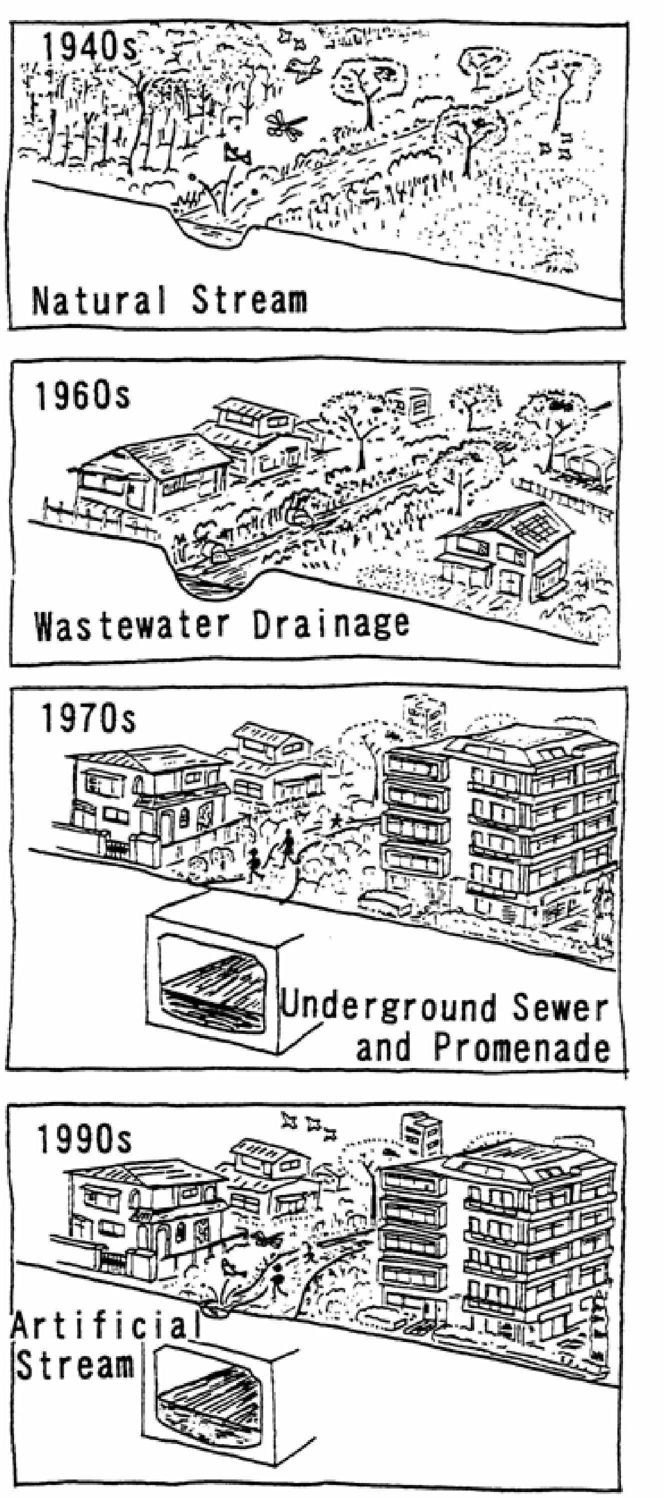

Figure 2-1. The history of Kitazawa Stream (Fujita 1998).

Another Tokyo example of urban stream rehabilitation has occurred in the Nogawa Stream watershed. The watershed is about 70 km^2 in area and has a population of about 700,000 people. Urbanization in this area also dramatically decreased the natural groundwater recharge to the stream. With development, household graywater, some sanitary wastewater, and stormwater were infiltrated into the ground and recharged the stream. When the sanitary wastewater collection and treatment system was improved in the 1980s, the stream flow was severely diminished, as a major source of groundwater recharge was eliminated. The headwater springs in the Nogawa area were of special importance to the local residents and they requested that TMG restore the dried springs. Artificial groundwater recharge, using stormwater, has been successfully used to restore the springs. Many private homes have installed stormwater infiltration devices in the area. In an example in Mitaka City, 4,000 infiltration "soakaways" were constructed during the three years from 1992 to 1995, allowing about 240,000 m^3/yr of stormwater to be infiltrated to revitalize the spring at Maruike. Koganei City residents installed more than 26,000 soakaways and 10.4 km of infiltration trenches at 5,700 homes (about 25% of all of the homes in the area). Other cities in the area have also helped residents install several thousand additional infiltration facilities. Spring flows have increased, although quantitative estimates are not yet available.

Fujita (1998) repeatedly states the great importance that the Japanese place on nature, especially flowing water and the associated landscaping and attracted animals. They are therefore willing to perform what seems to be extraordinary efforts in urban stream recovery programs in the world's largest city. The stream recovery program is but one element of the TMG's efforts to provide a reasonably balanced urban water program. Water reuse and conservation are important elements in their efforts. Stormwater infiltration to recharge groundwaters and the use of treated wastewaters for beneficial uses (including the above described stream restoration, plus landscaping irrigation, train washing, sewer flushing, fire fighting, etc.) are all important elements of these efforts, although this reuse currently only amounts to about 7% of the total annual water use in Tokyo.

Guidelines for the Reuse of Stormwater in Urban Areas

An obviously important consideration when examining the reuse of stormwater is the different quality requirements for the different reuse activities. Reuse guidelines are relatively rare, but Table 2-1 presents some guidance from Japan (Fujita 1998). The most serious restrictions relate to ensuring the safety of the water during inadvertent human contact. The prevention of nuisance conditions is also of concern.

Table 2-1. Quality Standards for the Reuse of Treated Wastewater in Japan (Fujita 1998)[1]

	Toilet Flushing	Fire Sprinklers	Landscape Irrigation	Recreation Use
Total Coliforms (MPN/100 mL)	<1,000	<50	<1,000	<50
Residual Chlorine (mg/L)	present	>0.4		
Color (Pt units)	No unpleasant appearance	No unpleasant appearance	<40	<10
Turbidity (NTU)	No unpleasant appearance	No unpleasant appearance	<10	<5
BOD_5 (mg/L)	<20	<20	<10	<3
Odor	Not unpleasant	Not unpleasant	Not unpleasant	Not unpleasant
pH	5.8 – 8.6	5.8 – 8.6	5.8 – 8.6	5.8 – 8.6

[1]In addition, the objectives for carp and crustaceans in urban streams include the following: total BOD_5: <8 mg/L; a water depth of at least 10 cm, and a stream velocity of at least 0.1 m/s.

Table 2-2 shows Maryland's reuse guidelines, along with acceptable use categories and per capita requirements (Mallory 1973). Only a small fraction (<10%) of the total residential water use requirements need to be of the highest quality water. Class AA water meets all U.S. Public Health Service Drinking Water Standards, class A water is very similar, except for taste and odor considerations, class B water has less restrictions, especially with respect to suspended solids, and class C water only has minimum requirements pertaining to corrosivity. All of these waters require disinfection by the state of Maryland. It is not likely that stormwater would be used for class AA uses without conventional water treatment, but lower levels of use may be feasible. Table 2-3 shows the specific maximum concentrations allowed for each reuse category, as determined by the state of Maryland, in addition to typical residential area stormwater quality. Average stormwater concentrations are presented, as needed storage would provide equalization of concentrations over short periods of time.

Table 2-2. Distribution of Maryland Residential Water Use and Required Quality (Mallory 1973)

Class	Use	Rate of Use (gal/person/day)	Percentage of Total Water Use
AA	Consumption by humans, food preparation, general kitchen use	6.5	7
A	Bathing, laundering, auto washing	31.0	36
B	Lawn irrigation	518 gal/day/acre	29
C	Toilet flushing	24.0	28

Table 2-3. Maximum Concentrations Allowed by Maryland for Different Reuse Categories, Compared to Typical Residential Stormwater Runoff (Mallory 1973)

Constituent (mg/L)	AA	A	B	C	Typical average residential stormwater quality and highest use without treatment (various references)
Total solids	150	500	500	1500	250 (A)
Suspended solids	-	-	10	30	50 (none)
Turbidity (NTU)	0-3	3-8	8-15	15-20	25 (none)
Color (color units)	15	20	30	30	25 (B)
pH (pH units)	7	6	6	6	6 to 9 (AA)
Oxygen, dissolved (minimum)	5	5	4	4	Near saturation (AA)
Total coliform bacteria (MPN/100 mL)	1	70	240	240	>10,000 (none)
Ammonia (as NH_3)	0.5	0.5	0.5	0.5	<0.1 (AA)
Nitrate (as NO_3)	45	50	50	50	1 (AA)
Phosphates	1	1	1	1	0.5 (AA)
Calcium	0.5	75	75	75	10 (A)
Chloride	50	250	250	250	<50 (AA)
Fluoride	1.5	3	3	3	0.03 (AA)
Iron	0.1	0.3	0.3	0.3	
Magnesium	0.5	150	150	150	1 (A)
Manganese	0.05	0.1	0.5	0.5	
Sulfate	50	200	400	400	10 (AA)
Arsenic	0.01	0.05	0.05	0.05	<0.05 (A)
Chromium (+6)	0.05	0.05	0.05	0.05	<0.05 (AA)
Copper	1.0	1	1.5	1.5	0.05 (AA)
Cyanide	0.01	0.2	0.2	0.2	0.05 (A)
Lead	0.05	0.1	0.1	0.1	0.05 (AA)
Zinc	5	15	15	15	0.5 (AA)

As shown on these tables, residential area stormwater can be used to meet at least class A water needs, except for suspended solids, turbidity, color, and coliform bacteria. The solids, turbidity and color levels are likely to be adequately reduced through storage and associated settling, plus possible post-settling filtration. The most serious impediment for the reuse of stormwater in residential areas are the bacteria levels. Unfortunately, stormwater is known to contain pathogens that can cause illness through various exposure mechanisms. However, it must be remembered that stormwater currently comes in contact with many people during rains and runoff from roofs and paved areas are encouraged to drain to landscaped areas to reduce runoff quantities. These practices are not considered hazardous and have not shown detrimental effects. Never-the-less, total coliform bacteria levels in stormwater can be very large, much greater than 10,000 MPM/100 mL and greatly exceed reuse criteria. The criteria for reuse shown on Table 2-3 requires a maximum total coliform level of 240 MPM/100 mL for class B and C water, and a level of 70 MPM/100 mL for class A water. Drinking water (class AA water) requires a maximum of 1 MPM/100 mL. Any of these levels would be impossible to meet without significant disinfection efforts.

Another set of reuse guidelines has been developed in California and are shown on Table 2-4. These guidelines were developed for the reuse of high quality secondary domestic wastewater effluent. The median total coliform bacteria criteria are very stringent (to product the public from likely associated pathogens) and would also not be possible to be met without very significant disinfection efforts. The only uses where primary treatment alone (similar to detention) is needed, and for which no total coliform bacteria criteria are given, are for the irrigation of fodder crops, fiber crops, seed crops, and for surface irrigation of processed produce. As indicated in Table 2-4, irrigation in areas where public contact is likely requires disinfection and very low levels of total coliform bacteria.

Table 2-4. California Reuse Guidelines (Metcalf and Eddy 1991)

Use of reclaimed water	Secondary treatment and disinfection	Secondary treatment, coagulation, filtration, and disinfection	Total coliform bacteria criteria (MPN/100 mL, median of daily observations)
Landscaped areas: golf courses, cemeteries, freeways	required		23
Landscaped areas: parks, playgrounds, schoolyards		required	2.2

Recreational impoundments: no public contact	required		23
Recreational impoundments: boating and fishing only	required		2.2
Recreational impoundments: body contact (bathing)		required	2.2

Metcalf and Eddy (1991) state that primary treatment (similar to settling in a storage tank) reduces fecal coliform bacteria by less than 10%, whereas trickling filtration (without disinfection) can reduce fecal coliform levels by 85 to 99%. Chemical disinfection is usually required to reduce pathogen levels by 99.9+%, as likely needed to meet the above bacteria criteria for even the most basic water uses. Because of the risks associated with potential pathogens, reuse of stormwater in residential areas should only be considered where consumption and contact is minimized, restricting on-site reuse to classifications B and C, and only after adequate disinfection and site specific study to ensure acceptable risks. To further minimize risks, only the best quality stormwater (from a pathogen perspective) should be considered for reuse. As an example, residential area roof runoff generally has lower fecal coliform concentrations than runoff from other source areas, although very high levels are periodically observed from this source area. Therefore, stormwater "harvesting" efforts could be limited to residential area rooftops to reduce risks associated with pathogens. The following subsection explores this example of reuse.

The Urban Water Budget and Stormwater Reuse in Residential Areas

Developing an urban water budget is the initial step needed when examining potential beneficial uses of stormwater. The urban water budget comprises many elements, stormwater being just one. As an example, it is possible to determine the likelihood of supplying needed irrigation water and toilet flushing water (reuse classifications B and C) from the stormwater generated from roof runoff by conducting an urban water budget. This budget requires a knowledge of all water sources and uses, and the associated quality requirements. Another important element is understanding the timing of the water needs and supplies. For example, the following lists household water use for a typical home (2 working adults and one child) in the southeast, where the rainfall averages about 50 inches per year:

- bathing 42%
- laundry 11%
- kitchen sink 15%
- dishwasher 8%
- bath sinks 12%
- toilet flushing 12%

Because this was a working family and the child was in school, bathing water use was relatively high, while the toilet flushing water use was relatively low. There were also wide variations in water use for different days of the week, with weekday water use (especially toilet flushing and laundry) being substantially less than for weekend water use. The household water use was relatively constant throughout the year and averaged about 90 gpcd (gal/capita/day), ranging from 77 to 106 gpcd. There were no water conservation efforts employed during the two year observation period. Outside irrigation water use during the dry months averaged about 50 gallons per day (for a ½ acre landscaped area) above the inside water uses listed above. Landscape irrigation may occur for about 2 months at this level of use in this area.

The estimated roof runoff for a typical 2,000 ft^2, 1- ½ level, house (roof area of about 1300 ft^2) would be about 40,000 gallons per year, for this area having about 50 inches of rain a year. The total water use for this household is about 100,000 gallons per year, with the amount used for toilet flushing being about 12,000 gallons, with another 3,000 gallons used for landscaping irrigation. For this example, the roof runoff would supply almost three times the amount of water needed for toilet flushing and landscape irrigation. None of the other household water uses would be suitable for supply by roof runoff. The rainfall varies between about 3 to 5 inches per month, with a rain occurring about twice a week on the average. Rainfall only once every two weeks can occur during the most unusual conditions (the driest months when landscaping irrigation is most needed). Therefore, a simple estimate for required roof runoff storage would be two weeks for average toilet flushing (450 gallons), plus two weeks for maximum landscaping irrigation (700 gallons). A total storage tank of 1250 gallons (a typical septic tank size) would therefore be needed. Of course, a factor-of-safety multiplier can be applied, depending on the availability of alternative water sources.

For a typical 0.5 acre residential lot in the southeast, the annual stormwater generated would be about 170,000 gallons per year. The roof would produce about 25% of this total, pavement would produce another 25%, and the landscaped area would produce about 50% of this total. Therefore, the amount of stormwater used on-site for toilet flushing and irrigation of landscaped areas would be only about 10% of the total generated. Therefore, most of the runoff would still have to be infiltrated on-site, or safely conveyed and discharged.

Other locations would obviously result in different water needs that could be supplied by runoff, depending on rainfall, soil conditions, and household water use patterns. Mitchell, *et al*. (1996) reported that on-site graywater and rain storage for re-use resulted in about 45% reductions in imported water needs, about 50% reductions in stormwater runoff, and about 10% reductions in wastewater discharges at two test developments in Australia. In most areas, Heaney, *et al*. (1998) reports that indoor water use is relatively constant at about 60 gpcd, with conservation practices, especially the use of low-flush toilets, possibly reducing this need to about 35 to 40 gpcd. Toilet flushing is about 30% of this use. In the arid parts of the U.S., landscaping irrigation can be the most important use of domestic water.

Heaney, *et al.* (1998) also reported the results of using water demand models to estimate the fraction of typical household irrigation water needs that could be satisfied by storing and using stormwater. Most eastern and west coast areas were able to satisfy their irrigation needs by storing stormwater for use on-site. Over 90% of the irrigation needs could be satisfied by stormwater re-use in the Rocky Mountain area and in the semi-arid southwest. The desert southwest was only able to supply about 25% of their irrigation needs with stormwater. Either supplemental irrigation, or the more appropriate selection of landscaping plants, would therefore be needed in these desert areas. Storage tank sizes varied widely and were quite large. Central Texas (San Antonio) required the largest tank size (25,000 gallons), while most of the eastern areas of the U.S. required less than 5,000 gallon tanks.

There are many areas that benefit from using poor quality water. A review by Paret and Elsner (1993) reported that some Florida golf courses use about 2,000 gal per acre per day of reclaimed sanitary wastewater. Other major Florida users of reclaimed sanitary wastewater include agricultural, horticultural and commercial users at about 1,500 gal per acre per day, and multifamily residential developments using about 3,000 gal per acre per day. The service fees for this reclaimed water ranged from about $0.05 to $0.64 per 1,000 gallons. Obviously, stormwater could be used for similar purposes, if stored and adequately treated. As an example, several new Veterans Affairs hospitals in the Los Angeles area are heavily landscaped using wet detention ponds holding stormwater tied into their fire fighting systems.

Besides on-site reuse of stormwater, dual distribution systems may be a feasible choice for many conditions. A dual water supply system includes a conventional domestic water supply system carrying class AA water for human consumption and bathing. Another water supply system is also used in a dual system carrying water of a lesser quality. This water is typically used for B and C uses, plus fire fighting. In areas having dual distribution systems, the poorer quality water is typically secondary sewage effluent that has received additional treatment, as noted above. Okun (1990) states that "throughout the world, dual distribution systems are proliferating, speeded up by policies adopted by states in the U.S. and governments elsewhere." He points out that a common feature of these water reuse/dual distribution systems is that customers pay for the reclaimed water, but at a significantly reduced price, compared to typical domestic water. He concluded that a sustainable wastewater reclamation program can only exist with cost recovery.

Even though most of the examples of dual distribution systems and wastewater reclamation are for sanitary wastewater, stormwater may be a much preferable degraded water source for reclamation (NAS 1994). Stormwater does not require nearly as high of a level of treatment, but it is not conveniently collected at one location such as at a wastewater treatment plant, nor is it available at such a constant and predicable flow as sanitary wastewater. However, the large volumes available and its generally better quality may make stormwater a more feasible water for dual distribution systems in many situations.

The Need for Change in Urban Water Management

As indicated above, stormwater can be considered a valuable resource in urban areas, not just a waste that must be rapidly discarded. Many have recognized this potential resource, as briefly outlined above. The *Symposium on Water, the City, and Urban Planning* was held in Paris, France, on April 10 and 11, 1997. The 300 participants formulated the *Paris Statement* outlining needed changes in urban water management. Even though stormwater management is usually considered a luxury of the developed countries (especially North America, Western Europe, and a few major Asian cities), this symposium stressed the need for recognizing the important role that stormwater management can play in the developing countries. Some of the major points of the *Paris Statement* are briefly outlined below:

- The marked process of urbanization in most countries, and especially in the developing world, is causing very rapid increases in water demands, often far outstripping available resources. Water management needed for sustainable urban development, let alone long-term survival of cities, requires immediate attention.

• Water related problems are affected by all elements of the water cycle, including water, land, air, and energy. Social, cultural, political, institutional, and economic aspects are integral and may even be dominant components of urban water management issues. Therefore, an integrated approach for solving urban water resource problems is necessary.

• Each city has a unique set of conditions and problems that require site specific solutions. However, a great deal of information from cities throughout the world is available for helping to solve these local problems.

• Demand management measures to encourage water conservation needs to be implemented, along with the timely consideration of environmentally sound projects to increase the availability of water when and where it is needed. Water problems are recognized mostly as temporal and spatial distribution problems, not because there is a fundamental shortage of water.

• An integrated management approach to surface and groundwaters is needed. Groundwater contamination by urban wastes must be controlled and safe recharge of groundwaters by wastewater and stormwater needs to be investigated.

• Appropriate approaches for urban drainage must consider variations in local climate, types of problems, and economic and maintenance capabilities. In addition, non-structural solutions need to be implemented as part of an integral approach to flood control in urban areas.

• There is a great need to conceive and apply new innovative solutions to solve urban water resource problems. This is especially likely and needed in areas with little drainage and sanitation infrastructure currently in place.

• The symposium recommended the creation of a single and integrated entity for coordination and management of water resources in each urban area.

Numerous papers were presented at the Engineering Foundation/ASCE sponsored symposium on *Sustaining Urban Water Resources in the 21st Century*, held in Malmo, Sweden, in September 1997, describing many international examples of effective urban water resources management. Sulsbrück and Forvaltning (1998) describe renovations being made to the drainage systems in Hillerød, Denmark. The town has 34,000 inhabitants, with about 600 mm or rainfall per year. The receiving water streams are quite small, being about 1 to 3 m across and have an annual average flow of about 600 L/s. About 3.5 km^2 of the drainage area has separate sanitary and storm sewers, while about 12.5 km^2 has combined sewers. The average dry weather flow to the treatment plant is about 14,000 m^3/day, and about 5,000 to 6,000 m^3 per day is lost to infiltration through leaky sewers. The amount lost through infiltration is about equal to the annual stormwater flow. Major sewer renovations are occurring to correct the leaking sewers and to minimize CSOs. Residential roof runoff is required to be infiltrated in newly developing areas, unless building moisture problems prevent its use. Industrial area runoff in new areas is directed to separate storm sewers, and detention facilities are being built to reduce stormwater flows to the streams to a maximum of 0.6 to 1 L/s/ha of drainage area. The sizes of the detention ponds range from 500 m^2 to 65,000 m^2. The total capacity of the retention ponds were 60,000 m^3 in 1997, with an additional 15,000 m^3 planned. The volume of CSOs was about 470,00 m^3 in 1990 and is expected to decrease to about 130,000 m^3 by 2001. Residential area roof runoff is not considered to cause pollution problems to soil or groundwater, while roadway runoff is usually not allowed to be infiltrated because of contamination concerns. Infiltration trenches are being retro-fitted at private homes, with labor provided by unemployed workers, who are paid by the government. The trenches are designed for a 2-year return period storm, the same as the storm sewers. The trenches for a typical 150 m^2 home range from 6 m long for gravelly soil sites to 24 m long for silty soil sites and cost about US$2,000 to construct (for a typical 9 m trench). They found that the use of combined sewers with infiltration is comparable in cost and pollutant discharges with a separate stormwater system. However, the infiltration system dramatically improves groundwater conditions, especially with the repair of the leaky sewers. The local residents also have had a change in attitude towards stormwater management. Runoff is now regarded as a resource instead of a waste. Sulsbrück and Forvaltning (1998) state that “many small, fine, green oases have been provided at the detention pond sites for citizen enjoyment and as habitat for plants and animals.”

A paper presented by Geldof (1998) at the Malmo conference on *Sustaining Urban Water Resources in the 21st Century* described changes that are occurring in the Netherlands. He stated that Dutch urban surface waters tended to be neglected in the past because of their poor water quality. However, current thinking is stressing significant changes in urban water management that will decrease many current problems (such as leaking sanitary and combined sewerage, discharges caused by peak flows, groundwater elevation variations and subsidence, and eutrophic surface waters). Two main changes are being used: changes in the sewerage systems, and increased source controls with on-site reuse of stormwater. In the Netherlands, combined sewers serve about 75% of the urban areas and have a capacity for about 7 mm or rain. Overflows occur when the rainfall exceeds this amount (as often as ten times a year). Separate sewers have been mostly built since the 1970s and now serve most of the remaining urban land area. The separate sewers solved the combined sewer overflow problems, but surprisingly did little to improve the annual mass discharges of pollutants. With separate drainage systems, none of the stormwater is treated at the municipal wastewater treatment plant. In addition, inappropriate discharges of sanitary sewage to the storm sewers are periodically found from inadvertent connections. A new system, termed an "improved separate system", was therefore developed. This drainage system consists of separate sanitary and storm drainage, but they are cross-connected with one-way gate valves enabling some stormwater to enter the sanitary drainage and be treated at the municipal wastewater treatment facility. The one-way gate values prevent sanitary sewage from entering the storm drainage. Pressurized sanitary sewerage is also sometimes used, with pumps used to discharge appropriate amounts of stormwater into the sanitary sewage system. An important aspect of the improved separate system is that only the most contaminated stormwater enters the stormwater drainage system and then the sanitary wastewater collection system for conveyance to the treatment facility. The least contaminated stormwater (typically just the roof runoff) is infiltrated on site, or potentially also used for toilet flushing, laundry, or irrigation purposes. The improved separate systems typically have a conveyance capacity to handle a 4 mm rain, which is capable of directing about 75 to 90% of the paved area stormwater runoff to the treatment facilities. Geldolf reported that a surprising side effect of source control is that it tends to upgrade people's perception of stormwater: "it becomes a pleasure rather than a nuisance." He also reports that residents have even become competitive about how they can most effectively use stormwater on site.

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II. Stressor Categories and their Effects on Humans and Ecosystems

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"As for Paris, within the last few years, it has been necessary to move most of the mouths of the sewers down stream below the last bridge."

Victor Hugo, 1862

Effects of Runoff on Receiving Waters

Many studies have shown the severe detrimental effects of urban and agricultural runoff on receiving waters. These studies have generally examined receiving water conditions above and below a city, by comparing two parallel streams, or by comparing to an ecoregion reference. However, only a few studies have examined direct cause and effect relationships of runoff for receiving water aquatic organisms (Heaney and Huber 1984, Burton and Moore 1999, Werner, *et al*. 2000, Vlaming, *et al*. 2000, Bailey, *et al*. 2000, Wenholz and Cunkilton 1995). Chapter 4 presents several case studies representing the major approaches to assessing receiving water problems, while this chapter presents a review of the major stressor categories and summarizes their observed effects.

Indicators of Receiving Water Biological Effects and Analysis Methodologies

There are a number of useful, well-proven, tools that can detect adverse biological effects in receiving waters (see also Chapter 6). When these tools are used correctly and combined in the proper framework, they can used to identify runoff-related problems. Kuehne (1975) studied the usefulness of aquatic organisms as indicators of pollution. He found that invertebrate responses are indicative of pollution for some time after an event, but they may not give an accurate indication of the nature of the pollutants. In-stream fish studies were not been employed as biological indicators much before 1975, but that they are comparable in many ways to invertebrates as quality indicators and can be more easily identified. However, because of better information pertaining to invertebrates and due to their limited mobility, certain invertebrate species may be sensitive to minor changes in water quality. Fish can be highly mobile and cover large section of a stream, as long as their passage is not totally blocked by adverse conditions. Fish disease surveys were also used during the Bellevue, Washington, urban runoff studies as an indicator of water quality problems (Scott, *et al*. 1982; Pitt and Bissonnette 1984). McHardy, *et al.* (1985) examined heavy metal uptake in green algae (*Cladophora glomerata*) from urban runoff for use as a biological monitor of specific metals.

It is necessary to use a range of measurement endpoints to characterize ecosystem quality in systems that receive multiple stressors (Marcy and Gerritsen,1996; Foran, *et al*. 1998; Burton and Baird 2000). Dyer and White (1996) examined the problem of multiple stressors affecting toxicity assessments. They felt that field surveys rarely can be used to verify simple single parameter laboratory experiments. They developed a watershed approach integrating numerous databases in conjunction with *in-situ* biological observations to help examine the effects of many possible causative factors (see also Chapter 6).

The interactions of stressors such as suspended solids and chemicals can be confounding and easily overlooked. Ireland, *et al*. (1996) found that exposure to UV radiation (natural sunlight) increased the toxicity of PAH contaminated urban sediments to *C. dubia*. The toxicity was removed when the UV wavelengths did not penetrate the water column to the exposed organisms. Toxicity was also reduced significantly in the presence of UV when the organic fraction of the stormwater was removed. Photo-induced toxicity occurred frequently during low flow conditions and wet weather runoff and was reduced during turbid conditions.

Johnson, *et al*. (1996) and Herricks, *et al*. (1996a and 1996b) describe a structured tier testing protocol to assess both short-term and long-term wet weather discharge toxicity that they developed and tested. The protocol recognizes that the test systems must be appropriate to the time-scale of exposure during the discharge. Therefore, three time-scale protocols were developed, for intra-event, event, and long-term exposures. The use of standard whole effluent toxicity (WET) tests were found to over-estimate the potential toxicity of stormwater discharges.

The effects of stormwater on Lincoln Creek, near Milwaukee, WI, were described by Crunkilton, *et al.* (1996). Lincoln Creek drains a heavily urbanized watershed of 19 mi^2 that is about nine miles long. On-site toxicity testing was conducted with side-stream flow-through aquaria using fathead minnows, plus in-stream biological assessments, along with water and sediment chemical measurements. In the basic tests, Lincoln Creek water was continuously pumped through the test tanks, reflecting the natural changes in water quality during both dry and wet weather conditions. The continuous flow-through mortality tests indicated no toxicity until after about the 14^{th} day of exposure, with more than 80% mortality after about 25 days, indicating that short-term toxicity tests likely underestimate stormwater toxicity. The biological and physical habitat assessments supported a definitive relationship between degraded stream ecology and urban runoff.

Rainbow (1996) presented a detailed overview of heavy metals in aquatic invertebrates. He concluded that the presence of a metal in an organism cannot tell us directly whether that metal is poisoning the organism. However, if compared to concentrations in a suite of well-researched biomonitors, it may be possible to determine if the accumulated concentrations are atypically high, with a possibility that toxic effects may be present. The user should be cautious, however, when attempting to relate tissue concentrations to effects or with bioconcentration factors. Many metals are essential and/or regulated by organisms and their internal concentrations bear no relationship to the concentrations in surrounding waters or sediments.

A battery of laboratory and *in-situ* bioassay tests are most useful when determining aquatic biota problems (Burton and Stemmer,1988; Burton, *et al*. 1996; Chapter 6). The test series may include microbial activity tests, along with exposures of zooplankton, amphipods, aquatic insects, bivalves and fish. Indigenous microbial activity responses correlated well with in-situ biological and chemical profiles. Bascombe, *et al*. (1990) also reported on the use of *in-situ* biological tests, using an amphipod exposed for five to six weeks in urban streams, to examine urban runoff receiving water effects. Ellis, *et al.* (1991) examined bioassay procedures for evaluating urban runoff effects on receiving water biota. They concluded that an acceptable criteria for protecting receiving water organisms should not only provide information on concentration and exposure relationships for *in-situ* bioassays, but also consider body burdens, recovery rates, and sediment related effects.

During the Coyote Creek, San Jose, California, receiving water study, 41 stations were sampled in both urban and nonurban perennial flow stretches of the creek. Short and long-term sampling techniques were used to evaluate the effects of urban runoff on water quality, sediment properties, fish, macroinvertebrates, attached algae, and rooted aquatic vegetation (Pitt and Bozeman 1982).

Fish Kills and Advisories

Runoff impacts are sometimes difficult for many people to appreciate in urban and agricultural areas. Fish kills are the most obvious indication of water quality problems for many people. However, because receiving water quality is often so poor, the aquatic life in typical urban and agricultural receiving waters is usually limited in abundance and diversity, and quite resistant to poor water quality. Sensitive native organisms have typically been displaced, or killed, long ago, and it usually requires an unusual event to cause a fish fill (Figure 3-1). Ray and White (1979) stated that one of the complicating factors in determining fish kills related to heavy metals is that the fish mortality may lag behind the first toxic exposure by several days, and is usually detected many miles downstream from the discharge location. The actual concentrations of the water quality constituents that may have caused the kill could then be diluted beyond detection limits, making probable sources of the toxic materials impossible to determine in many cases.

Figure 3-1. Fish kill in Village Creek, Birmingham, AL, due to Dursban entering storm drainage during warehouse fire.

Heaney, *et al*. (1980) reviewed fish kill information reported to government agencies during 1970 to 1979. They found that less than three percent of the reported 10,000 fish kills were identified as having been caused by urban runoff. This is less than 30 fish kills per year nationwide. However, the cause of these 10,000 fish kills usually could not be identified. It is expected that many of these fish kills could have been caused by runoff, or a combination of problems that could have been worsened by runoff. For example, elevated nutrient loading causes eutrophication that may lead to dissolved oxygen deficits and subsequent fish kills. These events are exacerbated by natural stressors such as low flow conditions. More recent surveys have found nearly 30% of fish kills are attributable to runoff (Figure 3-2, EPA 1995).

Figure 3-2. Sources associated with fish kills (EPA 1995).

During the Bellevue, Washington, receiving water studies, some fish kills were noted in the unusually clean urban streams (Pitt and Bissonnette 1984). The fish kills were usually associated with inappropriate discharges to the storm drainage system (such as cleaning materials and industrial chemical spills) and not from "typical" urban runoff. However, as noted later, the composition of the fish in the urban stream was quite different, as compared to the control stream (Scott, *et al*. 1986).

Fish kill data have, therefore, not been a good indicator for identifying stressor categories or types. However, the composition of the fisheries and other aquatic life taxonomic indicators are sensitive indicators of receiving water problems in streams.

In addition to fish kills, a significant concern is the increasing number of fish advisories being issued by States across the nation (Figure 3-3, EPA 1995). The causes of fish contamination and fish kills vary, but runoff is a primary contributor.

Figure 3-3. US fish consumption advisories (EPA 1995).

Adverse Aquatic Life Effects Caused by Runoff

Aquatic organisms are sensitive indicators of water quality. There have been many studies that describe aquatic life impairments that may result from exposure to contaminated runoff and/or habitat degradation. The following section summarizes some of these studies, which are typical of urban and agricultural watersheds.

Klein (1979) studied 27 small watersheds having similar characteristics, but having varying land uses, in the Piedmont region of Maryland. During an initial phase of the study, they found definite relationships between water quality and land use. Subsequent study phases examined aquatic life relationships in the watersheds. The principal finding was that stream aquatic life problems were first identified with watersheds having imperviousness areas comprising at least 12 percent of the watershed. Severe problems were noted after the imperviousness quantities reached 30 percent.

Receiving water impact studies were also conducted in North Carolina by Lenat, *et al*. (1979), Lenat and Eagleson (1981) and Lenat, *et al*. (1981). The benthic fauna occurred mainly on rocks. As sedimentation increased, the amount of exposed rocks decreased, with a decreasing density of benthic macroinvertebrates. Data from 1978 and 1979 in five cities showed that urban streams were grossly polluted by a combination of toxicants and sediment. Chemical analyses, without biological analyses, would have underestimated the severity of the problems because the water column quality varied rapidly, while the major problems were associated with sediment quality and effects on macroinvertebrates. Macroinvertebrate diversities were severely reduced in the urban streams, compared to the control streams. The biotic indices indicated "very poor" conditions for all urban streams. Occasionally, high populations of pollutant tolerant organisms were found in the urban streams, but would abruptly disappear before subsequent sampling efforts. This was probably caused by intermittent discharges of spills or illegal dumpings of toxicants. Although the cities studied were located in different geographic areas of North Carolina, the results were remarkably uniform.

A major nonpoint runoff receiving water impact research program was conducted in Georgia (Cook, *et al*. 1983). Several groups of researchers examined streams in major areas of the state. Benke, *et al*. (1981) studied 21 stream ecosystems near Atlanta having watersheds of one to three square miles each and land uses ranging from 0 to 98 percent urbanization. They measured stream water quality but found little relationship between water quality and degree of urbanization. The water quality parameters also did not identify a major degree of pollution. In contrast, there were major correlations between urbanization and the number of species. They had problems applying diversity indices to their study because the individual organisms varied greatly in size (biomass). CTA (1983) also examined receiving water aquatic biota impacts associated with nonpoint sources in Georgia. They studied habitat composition, water quality, macroinvertebrates, periphyton, fish, and toxicant concentrations in the water, sediment, and fish. They found that the impacts of land use were the greatest in the urban basins. Beneficial uses were impaired or denied in all three urban basins studied. Fish were absent in two of the basins and severely restricted in the third. The native macroinvertebrates were replaced with pollution tolerant organisms. The periphyton in the urban streams were very different from those found in the control streams and were dominated by species known to create taste and odor problems.

Pratt, *et al.* (1981) used basket artificial substrates to compare benthic population trends along urban and nonurban areas of the Green River in Massachusetts. The benthic community became increasing disrupted as urbanization increased. The problems were not only associated with times of heavy rain, but seemed to be affected at all times. The stress was greatest during summer low flow periods and was probably localized near the stream bed. They concluded that the high degree of correspondence between the known sources of urban runoff and the observed effects on the benthic community was a forceful argument that urban runoff was the causal agent of the disruption observed.

Cedar swamps in the New Jersey Pine Barrens were studied by Ehrenfeld and Schneider (1983). They examined nineteen swamps subjected to varying amounts of urbanization. Typical plant species were lost and replaced by weeds and exotic plants in urban runoff affected swamps. Increased uptakes of phosphorus and lead in the plants were found. It was concluded that the presence of stormwater runoff to the cedar swamps caused marked changes in community structure, vegetation dynamics, and plant tissue element concentrations.

Medeiros and Coler (1982 and 1984) used a combination of laboratory and field studies to investigate the effects of urban runoff on fathead minnows. Hatchability, survival, and growth were assessed in the laboratory in flow-through and static bioassay tests. Growth was reduced to one half of the control growth rates at 60 percent dilutions of urban runoff. The observed effects were believed to be associated with a combination of toxicants.

The benthos in the upper reaches of Coyote Creek (San Jose, California) consisted primarily of amphipods and a diverse assemblage of aquatic insects (Pitt and Bozeman 1982). Together those groups comprised two-thirds of the benthos collected from the nonurban portion of the creek. Clean water forms were abundant and included amphipods (*Hyaella azteca*) and various genera of mayflies, caddisflies, black flies, crane flies, alderflies, and riffle beetles. In contrast, the benthos of the urban reaches of the creek consisted almost exclusively of pollution tolerant oligochaete worms (tubificids). Tubificids accounted for 97 percent of the benthos collected from the lower portion of Coyote Creek.

There were significant differences in the numbers and types of benthic organisms found in the Bellevue Urban Runoff Program (Pederson 1981; Perkins 1982; Richey, *et al*. 1981; Richey 1982; Scott, *et al*. 1982). Mayflies, stoneflies, caddisflies, and bettles were rarely observed in Kelsey Creek, but were quite abundant in Bear Creek. These organisms are commonly regarded as sensitive indicators to environmental degradation. As an example of a degraded aquatic habitat in Kelsey Creek, a species of clams (*Unionidae*) was not found in Kelsey Creek, but was found in Bear Creek. These clams are very sensitive to heavy siltation and unstable sediments. Empty clam shells, however, were found buried in the Kelsey Creek sediments indicating their previous presence in the creek and their inability to adjust to the changing conditions. The benthic organism composition in Kelsey Creek varied radically with time and place while the organisms were much more stable in Bear Creek.

Introduced fishes often cause radical changes in the nature of the fish fauna present in a given waterbody. In many cases, they become the dominant fishes because they are able to out-compete the native fishes for food or space, or they may possess greater tolerance to environmental stress. In general, introduced species are most abundant in aquatic habitats modified by man while native fishes tend to persist mostly in undisturbed areas. Such is apparently the case within Coyote Creek, San Jose, California (Pitt and Bozeman 1982).

Samples from the nonurban portion of the study area were dominated by an assemblage of native fish species such as hitch, three spine stickleback, Sacramento sucker, and prickly sculpin. Rainbow trout, riffle sculpin, and Sacramento squawfish were captured only in the headwater reaches and tributary streams of Coyote Creek. Collectively, native species comprised 89 percent of the number and 79 percent of the biomass of the 2,379 fishes collected from the upper reaches of the study area. In contrast, native species accounted for only seven percent of the number and 31 percent of the biomass of the 2,899 fishes collected from the urban reach of the study area.

Hitch was the most numerous native fish species present. Hitch generally exhibit a preference for quiet water habitat and are characteristic of warm, low elevation lakes, sloughs, sluggish rivers and ponds. Mosquitofish dominated the collections from the urbanized section of the creek and accounted for over two-thirds of the total number of fish collected from the area. This fish is particularly well-adapted to withstand extreme environmental conditions, including those imposed by stagnant waters with low dissolved oxygen concentrations and elevated temperatures. The second most abundant fish species in the urbanized reach of Coyote Creek, the fathead minnow, is equally well suited to tolerate extreme environmental conditions. The species can withstand low dissolved oxygen, high temperature, high organic pollution and high alkalinity. Often thriving in unstable environments such as intermittent streams, the fathead minnow can survive in a wide variety of habitats.

The University of Washington (Pederson 1981; Perkins 1982; Richey, *et al*. 1981; Richey 1982; Scott, *et al*. 1982) conducted a series of studies to contrast the biological and chemical conditions in urban Kelsey Creek with rural Bear Creek. The urban creek was significant degraded when compared to the rural creek, but still supported a productive, but limited and unhealthy salmonid fishery. Many of the fish in the urban creek, however, had respiratory anomalies. The urban creek was not grossly polluted, but flooding from urban developments has increased dramatically in recent years. These increased flows have dramatically changed the urban stream's channel, by causing unstable conditions with increased stream bed movement, and by altering the availability of food for the aquatic organisms. The aquatic organisms are very dependent on the few relatively undisturbed reaches. Dissolved oxygen concentrations in the sediments depressed embryo salmon survival in the urban creek. Various organic and metallic priority pollutants were discharged to the urban creek, but most of them were apparently carried through the creek system by the high storm flows to Lake Washington. The urbanized Kelsey Creek also had higher water temperatures (probably due to reduced shading) than Bear Creek. This probably caused the faster fish growth in Kelsey Creek.

The fish population in Kelsey Creek had adapted to its degrading environment by shifting the species composition from coho salmon to less sensitive cutthroat trout and by making extensive use of less disturbed refuge areas (Figure 3-4). Studies of damaged gills found that up to three-fourths of the fish in Kelsey Creek were affected with respiratory anomalies, while no cutthroat trout and only two of the coho salmon sampled in Bear Creek had damaged gills. Massive fish kills in Kelsey Creek and its tributaries were observed on several occasions during the project due to the dumping of toxic materials down the storm drains.

Figure 3-4. Average biomass of fish at rural Bear Creek and urban Kelsey Creek sampling stations, Bellevue, WI.

Urban runoff impact studies were conducted in the Hillsborough River near Tampa Bay, Florida, as part of the NURP program (Mote Marine Laboratory 1984). Plants, animals, sediment, and water quality were all studied in the field and supplemented by laboratory bioassay tests. Effects of salt water intrusion and urban runoff were both measured because of the estuarine environment. During wet weather, freshwater species were found closer to the Bay than during dry weather. In coastal areas, these additional natural factors make it even more difficult to identify the cause and effect relationships for aquatic life problems. During another NURP project, Striegl (1985) found that the effects of accumulated pollutants in Lake Ellyn (Glen Ellyn, Ill.) inhibited desirable benthic invertebrates and fish and increased undesirable phyotoplankton blooms. LaRoe (1985) summarized the off-site effects of construction sediment on fish and wildlife. He noted that physical, chemical, and biological processes all affect receiving water aquatic life.

The number of benthic organism taxa in Shabakunk Creek in Mercer County, New Jersey, declined from 13 in relatively undeveloped areas to four below heavily urbanized areas (Garie and McIntosh 1986 and 1990). Periphyton samples were also analyzed for heavy metals with significantly higher metal concentrations found below the heavily urbanized area than above.

The Wisconsin Depratment of Natural Resources, in conjunction with the USGS and the University of Wisconsin conducted side-stream fish bioassay tests in Lincoln Creek in Milwaukee (Figures 3-5 and 3-6) (Crunkilton, *et al*. 1996). They identified significant acute toxicity problems associated with intermediate-term (about 10 to 20 day) exposures to adverse toxicant concentrations in urban receiving streams, with no indication of toxicity for shorter exposures. These toxicity effects were substantially (but not completely) reduced through the removal of stormwater particulates using a typical wet detention pond designed to remove most all of the particles larger than 5 µm in size.

Figure 3-5. Installation of side-stream fish bioassay test facilities at Lincoln Creek, Milwaukee, WI.

Figure 3-6. Lincoln Creek side-stream fish bioassay test facilities nearing completion.

Observed Habitat Problems Caused by Runoff

Some of the most serious effects of urban and agricultural runoff are on the aquatic habitat of the receiving waters. Numerous papers already referenced found significant sedimentation problems in receiving waters. These habitat effects are in addition to the pollutant concentration effects. The major problems how sediment affect the aquatic habitat include silting of spawning and food production areas and unstable bed conditions (Cordone and Kelley 1961). Other major habitat destruction problems include rapidly changing flows and the absence of refuge areas to protect the biota during these flow changes. Removal of riparian vegetation can increase water temperatures and a major source of debris that are important refuge areas. The major source of these habitat problems is the increased discharge volumes and flow rates associated with stormwater in developing areas that cause significant enlargements and unstable banks of small and moderate sized streams (Figures 3-7 and 3-8). Other habitat problems are caused by attempts to “correct” these problems by construction of lined channels (Figures 3-9 and 3-10) or small drop structures which hinder migration of aquatic life and create areas for the accumulation of contaminated silt (Figure 3-11).

Figure 3-7. Creek blowout after initial significant Spring rains in newly developed area (WI DNR photo).

Figure 3-8. Unstable banks and trash along Five-Mile Creek, Birmingham, AL.

Figure 3-9. Lined embankment along Waller Creek, Austin, TX.

Figure 3-10. Lined channel in Milwaukee, WI.

Figure 3-11. Small drop structure obstruction in Lincoln Creek, Milwaukee, WI.

Schueler (1996) stated that channel geometry stability can be a good indicator of the effectiveness of stormwater control practices. He also found that once a watershed area has more than about 10 to 15% effective impervious cover, noticeable changes in channel morphology occur, along with quantifiable impacts on water quality, and biological conditions. Stephenson (1996) studied changes in streamflow volumes in South Africa during urbanization. He found increased stormwater runoff, decreases in the groundwater table, and dramatically decreased times of concentration. The peak flow rates increased by about two-fold, about half caused by increased pavement (in an area having only about 5% effective impervious cover), with the remainder caused by decreased times of concentration.

Brookes (1988) has documented many cases in the U.S. and Great Britain of stream morphological changes associated with urbanization. These changes are mostly responsible for habitat destruction that are usually the most significant detriment to aquatic life. In many cases, water quality improvement would result in very little aquatic life benefits if the physical habitat is grossly modified. The most obvious habitat problems are associated with stream "improvement" projects, ranging from removal of debris, to straightening streams, to channelization projects. Brookes (1988 and 1991) presents a number of ways to minimize habitat problems associated with stream channel projects, including stream restoration.

Wolman and Schick (1967) observed deposition of channel bars, erosion of channel banks, obstruction of flows, increased flooding, shifting of channel bottoms, along with concurrent changes in the aquatic life, in Maryland streams affected by urban construction activities. Robinson (1976) studied eight streams in watersheds undergoing urbanization and found that the increased magnitudes and frequencies of flooding, along with the increased sediment yields, had considerable impact on stream morphology (and therefore aquatic life habitat).

The aquatic organism differences found during the Bellevue Urban Runoff Program were probably most associated with the increased peak flows in Kelsey Creek caused by urbanization and the resultant increase in sediment carrying capacity and channel instability of the creek (Pederson 1981; Perkins 1982; Richey, *et al*. 1981; Richey 1982; Scott, *et al*. 1982).

Kelsey Creek had much lower flows than Bear Creek during periods between storms. About 30 percent less water was available in Kelsey Creek during the summers. These low flows may also have significantly affected the aquatic habitat and the ability of the urban creek to flush toxic spills or other dry weather pollutants from the creek system (Ebbert, *et al*. 1983; Prych and Ebbert undated).

Kelsey Creek had extreme hydrologic responses to storm. Flooding substantially increased in Kelsey Creek during the period of urban development; the peak annual discharges almost doubled in the last 30 years, and the flooding frequency also increased due to urbanization (Ebbert, *et al*. 1983; Prych and Ebbert undated). These increased flows in urbanized Kelsey Creek resulted in greatly increased sediment transport and channel instability.

The Bellevue studies (summarized by Pitt and Bissonnette 1984) indicated that very significant interrelationships between the physical, biological, and chemical characteristics of the urbanized Kelsey Creek system. The aquatic life beneficial uses were found to be impaired and stormwater conveyance was most likely associated with increased flows from the impervious areas in the urban area. Changes in the flow characteristics could radically alter the ability of the stream to carry the polluted sediments into the other receiving waters. If the stream power (directly related to sediment carrying capacity) of Kelsey Creek was reduced, then these toxic materials could be expected to be settled into its sediment, with increased effects on the stream's aquatic life. Reducing peak flows would also reduce the flushing of smaller fish and other aquatic organisms from the system.

Many recent studies on urban stream habitats and restoration efforts have been conducted, especially in the Pacific Northwest. In one example, May, *et al.* (1999) found that maintaining natural land cover offers the best protection for maintaining stream ecological integrity and that best management practices have been generally limited in their ability to preserve appropriate conditions for lowland salmon spawning and rearing streams. They found that Puget Sound watersheds having a 10% impervious cover (likely resulting in marginal in-stream conditions) maintained at least 50% forested cover.

Groundwater Impacts from Stormwater Infiltration

Recently there have been some nation-wide studies that have shown virtually every agricultural and urban watershed contains elevated levels of nutrients, pesticides, and other organic chemicals in surface and ground waters, sediments, and fish tissues (e.g., USGS 1999). Since groundwaters are widely used as a drinking water and irrigation source and recharge many surface water bodies, the implications of chemical contamination are quite serious.

Prior to urbanization, groundwater recharge resulted from infiltration of precipitation through pervious surfaces, including grasslands and woods. This infiltrating water was relatively uncontaminated. With urbanization, the permeable soil surface area through which recharge by infiltration could occur was reduced. This resulted in much less groundwater recharge and greatly increased surface runoff. In addition, the waters available for recharge generally carried increased quantities of pollutants. With urbanization, new sources of groundwater recharge also occurred, including recharge from domestic septic tanks, percolation basins (Figure 3-12) and industrial waste injection wells, and from agricultural and residential irrigation. Special groundwater contamination problems may occur in areas having Karst geology where surface waters can be easily and quickly directed to the subsurface (Figures 3-13 and 3-14). Of course, there are many less dramatic opportunities for stormwater to enter the groundwater, including areas of porous paver blocks (Figures 3-15 through 3-17), grass swales (Figures 3-18 and 3-19), infiltration trenches (Figure 3-20), biofiltration areas (Figure 3-21) and simply from runoff flowing across grass (Figure 3-22). Many of these infiltration practices are done to reduce surface water impacts associated with stormwater discharges. If the infiltration is conducted through surface soils (such as for grass swales and grass landscaped areas), then groundwater contamination problems are significantly reduced. However, if subsurface infiltration is practices (especially through the use of injection wells), then the risk of groundwater contamination for many stormwater pollutants substantially increases (Pitt, *et al.* 1994 and 1996).

Figure 3-12. Groundwater recharge basin in Long Island, NY, using stormwater (NY DEC photo).

Figure 3-13. Karst geology at an Austin, TX, roadcut showing major channeling opportunities for surface water to enter the Edwards Aquifer.

Figure 3-14. Public education roadside sign in Austin, TX, warning about sensitive recharge zone.

Figure 3-15. Paver blocks for on-site infiltration in Essen, Germany.

Figure 3-16. Paver blocks for emergency and utility vehicle access, Madison, WI (under construction).

Figure 3-17. Paver blocks for occasional access road, Seattle Science Center, WA.

Figure 3-18. Grass swale in residential area, Milwaukee, WI.

Figure 3-19. Grass swale in office park area, Milwaukee, WI.

Figure 3-20. Stormwater infiltration through infiltration trench, office park, Lake Oswego, OR.

Figure 3-21. Biofiltration in parking area (Center for Watershed Protection photo).

Figure 3-22. Infiltration through grassed areas.

The Technical University of Denmark (Mikkelsen, *et al.* 1996a and 1996b) has been involved in a series of tests to examine the effects of stormwater infiltration on soil and groundwater quality. They found that heavy metals and PAHs present little groundwater contamination threat, if surface infiltration systems are used. However, they express concern about pesticides which are much more mobile. Squillace, *et al*. (1996) along with Zogorski, *et al*. (1996) presented information concerning stormwater and its potential as a source of groundwater MTBE contamination. Mull (1996) stated that traffic areas are the third most important source of groundwater contamination in Germany (after abandoned industrial sites and leaky sewers). The most important contaminants are chlorinated hydrocarbons, sulfate, organic compounds, and nitrates. Heavy metals are generally not an important groundwater contaminant because of their affinity for soils. Trauth and Xanthopoulus (1996) examined the long-term trends in groundwater quality at Karlsruhe, Germany. They found that the urban land use is having a long-term influence on the groundwater quality. The concentration of many pollutants have increased by about 30 to 40% over 20 years. Hütter and Remmler (1996) describe a groundwater monitoring plan, including monitoring wells that were established during the construction of an infiltration trench for stormwater disposal in Dortmund, Germany. The worst case problem expected is with zinc, if the infiltration water has a pH value of 4.

The following paragraphs (summarized from Pitt, *et al*. 1994 and 1996) describe the stormwater pollutants that have the greatest potential of adversely affecting groundwater quality during inadvertent or intentional stormwater infiltration, along with suggestions on how to minimize these potential problems.

Nutrients. Groundwater contamination of phosphorus has not been as widespread, or as severe, as for nitrogen compounds. Nitrates are one of the most frequently encountered contaminants in groundwater. Whenever nitrogen-containing compounds come into contact with soil, a potential for nitrate leaching into groundwater exists, especially in rapid-infiltration wastewater basins, stormwater infiltration devices, and in agricultural areas. Nitrate has leached from fertilizers and affected groundwaters under various turf grasses in urban areas, including golf courses, parks and home lawns. Significant leaching of nitrates occurs during the cool, wet seasons. Cool temperatures reduce denitrification and ammonia volatilization, and limit microbial nitrogen immobilization and plant uptake. The use of slow-release fertilizers is recommended in areas having potential groundwater nitrate problems. The slow-release fertilizers include urea formaldehyde (UF), methylene urea, isobutylidene diurea (IBDU), and sulfur-coated urea. Residual nitrate concentrations are highly variable in soil due to soil texture, mineralization, rainfall and irrigation patterns, organic matter content, crop yield, nitrogen fertilizer/sludge rate, denitrification, and soil compaction. Nitrate is highly soluble (>1 kg/L) and will stay in solution in the percolation water, after leaving the root zone, until it reaches the groundwater.

Nitrate has a low to moderate groundwater contamination potential for both surface percolation and subsurface infiltration/injection practices because of its relatively low concentrations found in most stormwaters. However, if the stormwater nitrate concentration was high, then the groundwater contamination potential would also likely be high.

Pesticides. Pesticide contamination of groundwater can result from agricultural, municipal and homeowner use of pesticides for pest control and their subsequent collection in stormwater runoff. A wide range of pesticides and their metabolites have been found in watersheds, which include typical urban pesticides in agricultural areas and vice versa. This cross contamination of pesticides into areas they are not being used is attributed to atmospheric transport. Heavy repetitive use of mobile pesticides on irrigated and sandy soils likely contaminates groundwater. Some insecticides, fungicides and nematocides must be mobile in order to reach the target pest and hence, they generally have the highest contamination potential. Pesticide leaching depends on patterns of use, soil texture, total organic carbon content of the soil, pesticide persistence, and depth to the water table.

The greatest pesticide mobility occurs in areas with coarse-grained or sandy soils without a hardpan layer, having low clay and organic matter content and high permeability. Structural voids, which are generally found in the surface layer of finer-textured soils rich in clay, can transmit pesticides rapidly when the voids are filled with water and the adsorbing surfaces of the soil matrix are bypassed. In general, pesticides with low water solubilities, high octanol-water partitioning coefficients, and high carbon partitioning coefficients are less mobile. The slower moving pesticides have been recommended in areas of groundwater contamination concern. These include the fungicides iprodione and triadimefon, the insecticides isofenphos and chlorpyrifos and the herbicide glyphosate. The most mobile pesticides include: 2,4-D, acenaphthylene, alachlor, atrazine, cyanazine, dacthal, diazinon, dicamba, malathion, and metolachlor.

Pesticides decompose in soil and water, but the total decomposition time can range from days to years. Literature half-lives for pesticides generally apply to surface soils and do not account for the reduced microbial activity found deep in the vadose zone. Pesticides with a thirty-day half life can show considerable leaching. An order-of-magnitude difference in half-life results in a five- to ten-fold difference in percolation loss. Organophosphate pesticides are less persistent than organochlorine pesticides, but they also are not strongly adsorbed by the sediment and are likely to leach into the vadose zone, and the groundwater. Perhaps a greater concern that has recently emerged is the widespread prevalence of toxic pesticide metabolites (break-down products) that are not routinely analyzed. The ecological and human health significance of this is not presently known, but will be a future topic of investigation.

Lindane and chlordane have moderate groundwater contamination potentials for surface percolation practices (with no pretreatment) and for subsurface injection (with minimal pretreatment). The groundwater contamination potentials for both of these compounds would likely be substantially reduced with adequate sedimentation pretreatment. Pesticides have been mostly found in urban runoff from residential areas, especially in dry-weather flows associated with landscaping irrigation runoff.

Other Organics. The most commonly occurring organic compounds that have been found in urban groundwaters include phthalate esters (especially bis(2-ethylhexyl)phthalate) and phenolic compounds. Other organics more rarely found, possibly due to losses during sample collection, have included the volatiles: benzene, chloroform, methylene chloride, trichloroethylene, tetrachloroethylene, toluene, and xylene. PAHs (especially benzo(a)anthracene, chrysene, anthracene and benzo(b)fluoroanthenene) have also been found in groundwaters near industrial sites.

Groundwater contamination from organics, like from other pollutants, occurs more readily in areas with sandy soils and where the water table is near the land surface. Removal of organics from the soil and recharge water can occur by one of three methods: volatilization, sorption, and degradation. Volatilization can significantly reduce the concentrations of the most volatile compounds in groundwater, but the rate of gas transfer from the soil to the air is usually limited by the presence of soil water. Hydrophobic sorption onto soil organic matter limits the mobility of less soluble base/neutral and acid extractable compounds through organic soils and the vadose zone. Sorption is not always a permanent removal mechanism, however. Organic re-solubilization can occur during wet periods following dry periods. Many organics can be at least partially degraded by microorganisms, but others cannot. Temperature, pH, moisture content, ion exchange capacity of soil, and air availability may limit the microbial degradation potential for even the most degradable organic.

1,3-dichlorobenzene may have a high groundwater contamination potential for subsurface infiltration/injection (with minimal pretreatment). However, it would likely have a lower groundwater contamination potential for most surface percolation practices because of its relatively strong sorption to vadose zone soils. Both pyrene and fluoranthene would also likely have high groundwater contamination potentials for subsurface infiltration/injection practices, but lower contamination potentials for surface percolation practices because of their more limited mobility through the unsaturated zone (vadose zone). Others (including benzo(a)anthracene, bis (2-ethylhexyl) phthalate, pentachlorophenol, and phenanthrene) may also have moderate groundwater contamination potentials, if surface percolation with no pretreatment, or subsurface injection/infiltration is used. These compounds would have low groundwater contamination potentials if surface infiltration was used with sedimentation pretreatment. Volatile organic compounds (VOCs) may also have high groundwater contamination potentials if present in the stormwater (likely for some industrial and commercial facilities and vehicle service establishments). The other organics, especially the volatiles, are mostly found in industrial areas. The phthalates are found in all areas. The PAHs are also found in runoff from all areas, but they are in higher concentrations and occur more frequently in industrial areas.

Pathogenic Microorganisms. Viruses have been detected in groundwater where stormwater recharge basins are located short distances above the aquifer. Enteric viruses are more resistant to environmental factors than enteric bacteria and they exhibit longer survival times in natural waters. They can occur in potable and marine waters in the absence of fecal coliforms. Enteroviruses are also more resistant to commonly used disinfectants than are indicator bacteria, and can occur in groundwater in the absence of indicator bacteria.

The factors that affect the survival of enteric bacteria and viruses in the soil include pH, antagonism from soil microflora, moisture content, temperature, sunlight, and organic matter. The two most important attributes of viruses that permit their long-term survival in the environment are their structure and very small size. These characteristics permit virus occlusion and protection within colloid-size particles. Viral adsorption is promoted by increasing cation concentration, decreasing pH and decreasing soluble organics. Since the movement of viruses through soil to groundwater occurs in the liquid phase and involves water movement and associated suspended virus particles, the distribution of viruses between the adsorbed and liquid phases determines the viral mass available for movement. Once the virus reaches the groundwater, it can travel laterally through the aquifer until it is either adsorbed or inactivated.

The major bacterial removal mechanisms in soil are straining at the soil surface and at intergrain contacts, sedimentation, sorption by soil particles, and inactivation. Because of their larger size than for viruses, most bacteria are therefore retained near the soil surface due to this straining effect. In general, enteric bacteria survive in soil between two and three months, although survival times up to five years have been documented.

Enteroviruses likely have a high groundwater contamination potential for all percolation practices and subsurface infiltration/injection practices, depending on their presence in stormwater (likely if contaminated with sanitary sewage). Other pathogens, including *Shigella*, *Pseudomonas aeruginosa*, and various protozoa, would also have high groundwater contamination potentials if subsurface infiltration/injection practices are used without disinfection. If disinfection (especially by chlorine or ozone) is used, then disinfection byproducts (such as trihalomethanes or ozonated bromides) would have high groundwater contamination potentials. Pathogens are most likely associated with sanitary sewage contamination of storm drainage systems, but several bacterial pathogens are commonly found in surface runoff in residential areas.

Heavy Metals and other Inorganic Compounds. Heavy metals and other inorganic compounds in stormwater of most environmental concern, from a groundwater pollution standpoint, are chromium, copper, lead, nickel, and zinc. However, the majority of metals, with the consistent exception of zinc, are mostly found associated with the particulate solids in stormwaters and are thus relatively easily removed through sedimentation practices. Filterable forms of the metals may also be removed by either sediment adsorption or are organically complexed with other particulates.

In general, studies of recharge basins receiving large metal loads found that most of the heavy metals are removed either in the basin sediment or in the vadose zone. Dissolved metal ions are removed from stormwater during infiltration mostly by adsorption onto the near-surface particles in the vadose zone, while the particulate metals are filtered out near the soil surface. Studies at recharge basins found that lead, zinc, cadmium, and copper accumulated at the soil surface with little downward movement over many years. However, nickel, chromium, and zinc concentrations have exceeded regulatory limits in the soils below a recharge area at a commercial site. Elevated groundwater heavy metal concentrations of aluminum, cadmium, copper, chromium, lead, and zinc have been found below stormwater infiltration devices where the groundwater pH has been acidic. Allowing percolation ponds to go dry between storms can be counterproductive to the removal of lead from the water during recharge. Apparently, the adsorption bonds between the sediment and the metals can be weakened during the drying period.

Similarities in water quality between runoff water and groundwater has shown that there is significant downward movement of copper and iron in sandy and loamy soils. However, arsenic, nickel, and lead did not significantly move downward through the soil to the groundwater. The exception to this was some downward movement of lead with the percolation water in sandy soils beneath stormwater recharge basins. Zinc, which is more soluble than iron, has been found in higher concentrations in groundwater than iron. The order of attenuation in the vadose zone from infiltrating stormwater is: zinc (most mobile) > lead > cadmium > manganese > copper > iron > chromium > nickel > aluminum (least mobile).

Nickel and zinc would likely have high groundwater contamination potentials if subsurface infiltration/injection were used. Chromium and lead would have moderate groundwater contamination potentials for subsurface infiltration/injection practices. All metals would likely have low groundwater contamination potentials if surface infiltration were used with sedimentation pretreatment.

Salts. Salt applications for winter traffic safety is a common practice in many northern areas and the sodium and chloride, which are collected in the snowmelt, travel down through the vadose zone to the groundwater with little attenuation. Soil is not very effective at removing salts. Salts that are still in the percolation water after it travels through the vadose zone will contaminate the groundwater. Infiltration of stormwater has led to increases in sodium and chloride concentrations above background concentrations. Fertilizer and pesticide salts also accumulate in urban areas and can leach through the soil to the groundwater.

Studies of depth of pollutant penetration in soil have shown that sulfate and potassium concentrations decrease with depth, while sodium, calcium, bicarbonate, and chloride concentrations increase with depth. Once contamination with salts begins, the movement of salts into the groundwater can be rapid. The salt concentration may not decrease until the source of the salts is removed.

Chloride would likely have a high groundwater contamination potential in northern areas where road salts are used for traffic safety, irrespective of the pretreatment, infiltration or percolation practice used. Salts are at their greatest concentrations in snowmelt and early spring runoff in northern areas.

Stressor Categories and Their Effects

There are several ways in which stormwater stressors may be grouped. Overlap between these categories will occur since the ecosystem is comprised of interrelated, interactive components. Attempts at studying single stressors or single categories represents a “reductionist” approach as opposed to a more realistic “holistic” ecosystem approach (Chapman, *et al*. 1992). However, for one to understand the whole system and its response to stormwater stressors, there must first be a basic understanding of single component effects and patterns (see also Chapter X). The adverse effect of stormwater runoff has been mainly documented indirectly in NPS effect studies in urban and agricultural watersheds. The aquatic ecosystems in these environments typically show a loss of sensitive species, loss of species numbers (diversity and richness), and increases in numbers of pollution tolerant organisms (e.g., Schueler 1987, EPA 1987a, Pitt and Bozeman 1982, and Pitt 1995). These trends are observed at all levels of biological organization including fish, insects, zooplankton, phytoplankton, benthic invertebrates, protozoa, bacteria, and macrophytes. These changes tend to change an aquatic ecosystem from a stable system to an unstable one, and from a complex system to an overly simplistic one. As disturbances (e.g., toxic stormwater discharges) increase in frequency and severity, the recovery phase will increase and the ability to cope with a disturbance decrease (Figure 3-23). The following categories are but a generalized summary of commonly observed characteristics and effects in previous stormwater and ecotoxicological studies.

Figure 3-23. Concept of stable and unstable systems that divides time into phases for a system responding to a disturbance. Phases include reaction phase (A); recovery phase (R); persistence phase (P); and the time for disturbance recurrence (D). The stability of a system is recognized by ratios of recovery to disturbance recurrence (R:D). 1 and 2 indicate systems responding to disturbances of different magnitudes (Wissmar and Swanson 1990).

Stream Flow Effects and Associated Habitat Modifications

Some of the most serious effects of urban and agricultural runoff are on the aquatic habitat of the receiving waters. A major habitat destruction threat comes from the rapidly changing flows and the absence of refuge areas to protect the biota during these flow changes. The natural changes in stream hydrology will change naturally at a slow, relatively nondetectable rate in most areas of the United States, where streambanks are stabilized by riparian vegetation. In other areas, however, natural erosion and bank slumping will occur in response to high flow events. This “natural” contribution to stream solids is accelerated by hydromodifications, such as increases in stream power due to upstream channelization, installation of impervious drainage networks, increased impervious areas in the watershed (roof tops, roadways, parking areas), and removal of trees and vegetation. All of these increase the runoff volume and stream power, and decrease the time period for stream peak discharge.

In moderately developed watersheds, peak discharges are two to five times those of pre-development levels (Leopold 1968, Anderson 1970). These storm events may have 50% greater volume which may result in flooding. The quicker runoff periods reduce infiltration thus interflows and baseflows into the stream from groundwater during drought periods are reduced, as are groundwater levels. As stream power increases, channel morphology will change with an initial widening of the channel to as much as 2 to 4 times their original size (Robinson 1976, Fox 1974, Hammer 1972). Floodplains increase in size, stream banks are undercut and riparian vegetation lost. The increased sediment loading from erosion moves through the watershed as bedload, covering sand, gravel, and cobble substrates.

The aquatic organism differences found during the Bellevue Urban Runoff Program were probably most associated with the increased peak flows in Kelsey Creek caused by urbanization and the resultant increase in sediment carrying capacity and channel instability of the creek (Pederson 1981; Perkins 1982; Richey, *et al*. 1981; Richey 1982; Scott, *et al* . 1982). Kelsey Creek had much lower flows than Bear Creek during periods between storms. About 30 percent less water was available in Kelsey Creek during the summers. These low flows may also have significantly affected the aquatic habitat and the ability of the urban creek to flush toxic spills or other dry weather pollutants from the creek system (Ebbert, *et al*. 1983; Prych and Ebbert undated). Kelsey Creek had extreme hydrologic responses to storm. Flooding substantially increased in Kelsey Creek during the period of urban development; the peak annual discharges almost doubled in the last 30 years, and the flooding frequency also increased due to urbanization (Ebbert, *et al.* 1983; Prych and Ebbert undated). These increased flows in urbanized Kelsey Creek resulted in greatly increased sediment transport and channel instability.

The Bellevue studies (Pitt and Bissonnette 1984) indicated very significant interrelationships between the physical, biological, and chemical characteristics of the urbanized Kelsey Creek system. The aquatic life beneficial uses were found to be impaired and stormwater conveyance was most likely associated with increased flows from the impervious areas in the urban area. Changes in the flow characteristics could radically alter the ability of the stream to carry the polluted sediments into the other receiving waters.

Stephenson (1996) studied changes in streamflow volumes in South Africa during urbanization. He found increased stormwater runoff, decreases in the groundwater table, and dramatically decreased times of concentration. The peak flow rates increased by about two-fold, about half caused by increased pavement (in an area having only about 5% effective impervious cover), with the remainder caused by decreased times of concentration.

Bhaduri, *et al.* (1997) quantified the changes in streamflow and associated decreases in groundwater recharge associated with urbanization. They point out that the most widely addressed hydrologic effect of urbanization is the peak discharge increases that cause local flooding. However, the increase in surface runoff volume also represents a net loss in groundwater recharge. They point out that urbanization is linked to increased variability in volume of water available for wetlands and small streams, causing "flashy" or "flood-and-drought" conditions. In northern Ohio, urbanization at a study area was found to cause a 195% increase in the annual volume of runoff, while the expected increase in the peak flow for the local 100-yr event was 26% for the same site. Although any increase in severe flooding is problematic and cause for concern, the much larger increase in annual runoff volume, and associated decrease in groundwater recharge, likely has a much greater effect on in-stream biological conditions.

A number of presentations concerning aquatic habitat effects from urbanization were made at the *Effects of Watershed Development and Management on Aquatic Ecosystems* conference held in Snowbird, UT, in August of 1996, sponsored by the Engineering Foundation and the ASCE. MacRae (1997) presented a review of the development of the common zero runoff increase (ZRI) discharge criterion, referring to peak discharges before and after development. This criterion is commonly met using detention ponds for the 2 yr storm. MacRae shows how this criterion has not effectively protected the receiving water habitat. He found that stream bed and bank erosion is controlled by the frequency and duration of the mid-depth flows (generally occurring more often than once a year), not the bank-full condition (approximated by the 2 yr event). During monitoring near Toronto, he found that the duration of the geomorphically significant pre-development mid-bankfull flows increased by a factor of 4.2 times, after 34% of the basin had been urbanized, compared to before development flow conditions. The channel had responded by increasing in cross-sectional area by as much as 3 times in some areas, and was still expanding. Table 3-1 shows the modeled durations of critical discharges for predevelopment conditions, compared to current and ultimate levels of development with "zero runoff increase" controls in place. At full development and even with full ZRI compliance in this watershed, the hours exceeding the critical mid-bankfull conditions will increase by a factor of 10, with resulting significant effects on channel stability and the physical habitat.

MacRae (1997) also reported other studies that found that channel cross-sectional areas began to enlarge after about 20 to 25% of the watershed was developed, corresponding to about a 5% impervious cover in the watershed. When the watersheds are completely developed, the channel enlargements were about 5 to 7 times the original cross-sectional areas. Changes from stable streambed conditions to unstable conditions appear to occur with basin imperviousness of about 10%, similar to the value reported for serious biological degradation. He also summarized a study conducted in British Columbia that examined 30 stream reaches in natural areas, in urbanized areas having peak flow attenuation ponds, and in urbanized areas not having any stormwater controls. The channel widths in the uncontrolled urban streams were about 1.7 times the widths of the natural streams. The streams having the ponds also showed widening, but at a reduced amount compared to the uncontrolled urban streams. He concluded that an effective criterion to protect stream stability (a major component of habitat protection) must address mid-bankfull events, especially by requiring similar durations and frequencies of stream power (the product of shear stress and flow velocity, not just flow velocity alone) at these depths, compared to satisfactory reference conditions.

Table 3-1. Hours of Exceedence of Developed Conditions with Zero Runoff Increase Controls Compared to Predevelopment Conditions (MacRae (1997)

Recurrence Interval (yrs)	Existing Flowrate (m^3/s)	Exceedence for Predevelopment Conditions (hrs per 5 yrs)	Exceedence for Existing Development Conditions, with ZRI Controls (hrs per 5 yrs)	Exceedence for Ultimate Development Conditions, with ZRI Controls (hrs per 5 yrs)
1.01 (critical mid-bankfull conditions)	1.24	90	380	900
1.5 (bankfull conditions)	2.1	30	34	120

Urbanization radically affects many natural stream characteristics. Pitt and Bissonnette (1984) reported that the coho and cutthroat were affected by the increased nutrients and elevated temperatures of the urbanized streams in Bellevue, as studied by the University of Washington as part of the U.S. EPA's NURP project (EPA 1983). These conditions were probably responsible for accelerated growth of the fry which were observed to migrate to Puget Sound and the Pacific Ocean sooner than their counterparts in the control forested watershed that was also studied. However, the degradation of sediments, mainly the decreased particle sizes, adversely affected their spawning areas in streams that had become urbanized. Sovern and Washington (1997) reported that, in Western Washington, frequent high flow rates can be 10 to 100 times the predevelopment flows in urbanized areas, but that the low flows in the urban streams are commonly lower than the predevelopment low flows. They have concluded that the effects of urbanization on western Washington streams are dramatic, in most cases permanently changing the stream hydrologic balance by: increasing the annual water volume in the stream, increasing the volume and rate of storm flows, decreasing the low flows during dry periods, and increasing the sediment and pollutant discharges from the watershed. With urbanization, the streams increase in cross-sectional area to accommodate these increased flows and headwater downcutting occurs to decrease the channel gradient. The gradients of stable urban streams are often only about 1 to 2 percent, compared to 2 to 10 percent gradients in natural areas. These changes in width and the downcutting result in very different and changing stream conditions. The common pool/drop habitats are generally replaced by pool/riffle habitats, and the stream bed material is comprised of much finer material, for example. Along urban streams, fewer than 50 aquatic plant and animal species are usually found. They have concluded that once urbanization begins, the effects on stream shape are not completely reversible. Developing and maintaining quality aquatic life habitat, however, is possible under urban conditions, but it requires human intervention and it will not be the same as for forested watersheds.

Increased flows due to urban and agricultural modification obviously cause aquatic life impacts due to destroyed habitat (unstable channel linings, scour of sediments, enlarging stream cross-sections, changes in stream gradient, collapsing of riparian stands of mature vegetation, siltation, embeddedness, etc.) plus physical flushing of aquatic life from refuge areas downstream. The increases in peak flows, annual runoff amounts, and associated decreases in groundwater recharge obviously cause decreased dry weather flows in receiving streams. Many small and moderate-sized streams become intermittent after urbanization, causing extreme aquatic life impacts. Even with less severe decreased flows, aquatic like impacts can be significant. Lower flows are associated with increased temperatures, increased pollutant concentrations (due to decreased mixing and transport), and decreased mobility and forage opportunities.

Safety Concerns with Stormwater

There are many aspects of safety associated with urban and agricultural waters, including:

- Exposure to pathogens and toxicants
- Flows (rapidly changing and common high flows)
- Steep banks/cut banks/muddy/slippery banks
- Mucky sediments
- Debris (sharps and strainers)

- Habitat for nuisance organisms (muskrats, cottonmouths, “wild” dogs, etc.)

Most urban receiving waters having direct storm drainage outfalls are quite small and have no formally designated beneficial uses. Larger receiving waters typically have basic uses established, but few urban receiving waters have water contact recreation as a designated beneficial use. Unfortunately, these small waters typically attract local children where they may be exposed to some of the hazards associated with stormwater, as noted above. Conditions associated with pathogens and toxicants are likely a serious problem, but the other listed hazards are also very serious. Obviously, drowning should be a concern to all and is often a topic of heated discussion at public meetings where wet detention ponds for stormwater treatment are proposed. However, drowning hazards may be more common in typical urban streams than in well-designed wet detention ponds. These hazards are related to rapidly changing water flows, high flow rates, steep and muddy stream banks, and mucky stream deposits. These hazards are all increased with stormwater discharges and are typically much worse than in pre-development times when the streams were much more stable. This can be especially critical in newly developing areas where the local streams are thought to be relatively safe from prior experience, but rapidly degrade with increased development and associated stormwater discharges. Other potentially serious hazards are related to debris thrown into streams or trash dumped along stream banks. In unstable urban streams, streambanks often are continuously cut away, with debris (bankside trees, small buildings, trash piles, and even automobiles) falling into the waterway.

Many people also see untidy urban stream corridors as habitat for snakes and other undesirable creatures and like to clearcut the riparian vegetation and plant grass to the water’s edge. Others see creeks as convenient dumping grounds and throw all manner of junk (yard wastes, old appliances, etc.) over the back fences at homes or off bridges into stream corridors. Both of these approaches greatly hinder the use of streams. In contrast, residents of Bellevue, WA, have long accepted the value of small urban streams as habitat for fish. As an example, they have placed large amounts of gravel into streams to provide suitable spawning habitat. In other Northwest area streams, large woody debris are carefully placed into urban streams (using large street-side cranes, and sometimes even helicopters) to improve the aquatic habitat. In these areas, local residents are paying a great deal of money to improve the habitat along the streams and are obviously much more careful about creating hazards associated with junk and other inappropriate debris or discharges.

Drowning Hazards

Marcy and Flack (1981) state that drownings in general most often occur because of slips and falls into water, unexpected depths, cold water temperatures, and fast currents. Four methods to minimize these problems include: eliminate or minimize the hazard, keep people away, make the onset of the hazard gradual, and provide escape routes.

Jones and Jones (1982) consider safety and landscaping together because landscaping should be used as an effective safety element. They feel that appropriate slope grading and landscaping near the water’s edge can provide a more desirable approach than wide-spread fencing around wet detention ponds. Fences are expensive to install and maintain and usually produce unsightly pond edges. They collect trash and litter, challenge some individuals who like to defy barriers, and impede emergency access if needed. Marcy and Flack (1981) state that limited fencing may be appropriate in special areas. When the side slopes of a wet detention pond cannot be made gradual (such as when against a railroad right-of-way or close to a roadway), steep sides having submerged retaining walls may be needed. A chain link fence located directly on the top of the retaining wall very close to the water’s edge may be needed (to prevent human occupancy of the narrow ledge on the water side of the fence). Another area where fencing may be needed is at the inlet or outlet structures of wet detention ponds. However, fencing usually gives a false sense of security, as most can be easily crossed (Eccher 1991).

Common recommendations to maximize safety near wet detention ponds include suggestions that the pond side slopes be gradual near the water edge, with a submerged ledge close to shore. Aquatic plants on the ledge would decrease the chance of continued movement to deeper water and thick vegetation on shore near the water edge would discourage access to the water edge and decrease the possibility of falling into the water accidentally. Pathways should not be located close to the water's edge, or turn abruptly near the water. Marcy and Flack (1981) also encourage the placement of escape routes in the water whenever possible. These could be floats on cables, ladders, hand-holds, safety nets, or ramps. They should not be placed to encourage entrance into the water.

The use of inlet and outlet trash racks and antivortex baffles is also needed to prevent access to locations having dangerous water velocities. Several types are recommended by the NRCS (SCS 1982). Racks need to have openings smaller than about 6 inches to prevent people from passing through them and need to be placed where water velocities are less than three feet per second to allow people to escape (Marcy and Flack 1981). Besides maintaining safe conditions, racks also help keep trash from interfering with the operation of the outlet structure.

Eccher (1991) lists the following pond attributes to ensure maximum safety, while having good ecological control:

1) There should be no major abrupt changes in water depth in areas of uncontrolled access,
2) slopes should be controlled to insure good footing,
3) all slope areas should be designed and constructed to prevent or restrict weed and insect growth (generally requiring some form of hardened surface on the slopes), and
4) shoreline erosion needs to be controlled.

Obviously, many of these suggestions to improve safety near wet detention ponds may also be applicable to urban stream corridors. Of course, streams can periodically have high water velocities and steep banks may be natural. However, landscaping and trail placement along urban stream corridors can be carefully done to minimize exposure to the hazardous areas.

Aesthetics, Litter/Floatables, and other Debris Associated with Stormwater

One of the major problems with the aesthetic degradation of receiving waters in urban areas is a general lack of respect for the local water bodies. In areas where stormwater is considered a beneficial component of the urban water cycle, these problems are not as severe and inhabitants and visitors enjoy the local waterscape. The following list indicates the types of aesthetic problems that are common for neglected waters:

- Low flows
- Mucky and smelly sediments
- Trash from illegal dumping
- Floatables from discharges of litter
- Unnatural riparian areas
- Unnatural channel modifications
- Odiferous water and sediment
- Rotting vegetation and dead fish
- Objectionable sanitary wastes from CSOs and SSOs

The above list indicates the most obvious aesthetic problems in receiving waters. Many of these problems are directly associated with poor water quality (such as degraded sediments, eutrophication, and fish kills). Other direct problems associated with runoff include massive modifications of the hydrologic cycle with more severe and longer durations of low flow periods due to reduced infiltration of rainwater. Many of the other problems on the above list are related to indirect activities of the inhabitants of the watershed, namely illegal dumping of trash into streams, littering in the drainage area, and improper streambank modifications. In many areas, separate sewer overflows (SSOs) and combined sewer overflows (CSOs) also contribute unsightly and hazardous debris to urban receiving waters.

Floatable Litter Associated with Wet Weather Flows

As previously indicated, aesthetics is one of the most important beneficial uses recognized for urban waterways. Floatable litter significantly degrades the aesthetic enjoyment of receiving waters. The control of floatables has therefore long been a goal of most communities.

In coastal areas, land-based sources of beach debris and floatable material have generally been found to originate from wet weather discharges from the land, and not from marine sources (such as shipping). Of course, in areas where solid wastes (garbage or sewage sludge, for example) have been (or are still being) dumped in the sea, these sources may also be significant beach litter sources. In CSO areas, items of sanitary origin are found in the receiving waters and along the beaches, but stormwater discharges are responsible for most of the bulk litter material, including much of the hazardous materials. In inland areas, marine contributions are obviously not an issue. Therefore, with such direct linkages to the drainage areas, much of the floatable material control efforts have focused on watershed sources and controls (including being part of the “nine minimum” controls for CSOs required by the EPA). Figure 3-24 shows a schematic of how street and sidewalk litter enter the receiving waters (HydroQual 1995).

Figure 3-24. Schematic of transport of street and sidewalk litter into receiving waters (HydroQual 1995).

An example of an investigation of beach litter sources was conducted by Williams and Simmons (1997) along the Bristol Channel in the UK. They concluded that most of the litter accumulating on the beaches originated from river discharges, and not from litter being deposited directly on the beaches by visitors or from shipping or other oceanic sources. The sources of the litter into the major rivers were the many combined sewer overflows in the area. About 3,000 CSOs exist in Wales, and 86 of the 126 CSOs discharging into the study area receive no treatment. They reported previous studies that have concluded that about half of Britain's coastline is contaminated, with an average of 22 plastic bottles, 17 cans, and 20 sanitary items occurring per km of coast. In some areas, the beach litter can exceed 100 items per category per km. Their survey found that low energy (relatively flat) sandy beaches collected the most debris. Winter litter loadings were generally higher than during the summer, further indicating that storm related sources were more important than visitor related sources. They concluded that the linear strip development in South Wales' valleys had lead to rivers being used as open sewers and as general dumping grounds.

One of the largest and most comprehensive beach litter and floatable control investigations and control efforts in the U.S. has been conducted by New York City. At the beginning of their description of this floatable control program, Grey and Oliveri (1998) stated that "one of the major issues of urban wet weather pollution is the control of floatable pollution." The comprehensive New York City program included investigations of the sources of the litter contributing to the floatable discharges (mostly street and sidewalk litter) and the effectiveness of many floatable control practices (including public education, enhanced street cleaning, catchbasin hoods, floatable capture nets, and booming and skimmer boats) (Figures 3-25 through 3-28).

Figure 3-25. Trash boom, New York City, NY.

Figure 3-26. New York booms and skimmers for the control of floatable discharges.

Figure 3-27. TrashTrap™ at Fresh Creek, Brooklyn, NY.

Figure 3-28. New York City's use of end-of-pipe TrashTrap™ systems.

New York City used in-line net boxes installed below catchbasin inlets to capture the discharge of floatables for identification and quantification. Much of their work was directed at the capture efficiency of the floatable material in catchbasins. They found that it was critical that hoods (covers over the catchbasin outlets that extended below the standing water) be used in the catchbasins to help retain the captured material. They found that the hoods increased the capture of the floatables by 70 to 85%. Unhooded catchbasins were found to discharge about 11 grams per 100 ft of curb length per day, while hooded catchbasins reduced this discharge to about 3.3 grams per 100 ft of curb length per day. They also found that the hoods greatly extended the period of time (extending the cleaning interval) and the depth of accumulated litter that could be captured in the catchbasins without degraded capture performance.

There are about 130,000 stormwater inlet structures in New York City's 190,000 acres served by combined and separate sewers, or about 1.5 acres served by each inlet. They are surveying all of these inlet structures, replacing damaged or missing hoods, and accurately measuring their dimensions and indicating their exact locations for a city-wide GIS system. Catchbasin cleaning costs are about $170 per inlet, while the inspection and mapping costs are about $45 per inlet. Replacement hood costs are about $45 per inlet.

Litter surveys conducted by the New York City Department of Sanitation (DOS) in 1984 and 1986 found that 70% of the street litter items consisted of food and beverage wrappers and containers (60%) and the paper and plastic bags (10%) used to carry these items. The early studies also found that litter levels on the streets and sidewalks were about 20 to 25% higher in the afternoons than in the mornings. The DOS conducted similar surveys in 1993 at 90 blockfaces throughout the city (HydroQual 1995). Each litter monitoring site was monitored several times simultaneously when the surveys were conducted with the floatable litter separated into 13 basic categories. They found that twice as much floatable litter was located on the sidewalks compared to the streets (especially glass) and that land use had little effect on the litter loadings (except in the special business districts where enhanced street cleaning/litter control was utilized, resulting in cleaner conditions). Their baseline monitoring program determined that an average of 2.3 floatable litter items were discharged through the catchbasin inlets per day per 100 ft. of curb. This amount was equivalent to about 6.2 in^2 and 0.0134 lbs (8.5 grams) of material. The total litter load discharged was about twice this floatable amount. Table 3-2 summarizes the characteristics of the floatable litter found on the streets.

Table 3-2. Floatable Litter Characteristics found on New York City Streets (HydroQual 1995)

	# of items (%)	Weight of items (%)	Density of items (lb/ft^3)
Plastic	57.2	44.3	2.8
Metal	18.9	12.0	3.8
Paper (coated/waxed)	5.9	4.0	2.0
Wood	5.9	5.3	7.7
Polystyrene	5.4	1.3	0.7
Cloth/fabric	2.5	12.5	8.3
Sensitive items	1.7	0.4	na
Rubber	1.1	1.1	10.5
Misc.	1.0	3.6	9.8
Glass	0.4	15.6	13.8

Solids (Suspended, Bedded, and Dissolved)

The detrimental effects of elevated suspended and dissolved solids and increases in siltation and fine grained bedded sediments have been well-documented (EPA 1987b). The sources of these solids are primarily from dry deposition, roadways, construction, and channel alteration and have significant effects on receiving system habitats. Solids concentrations are directly related to watershed use characteristics and watershed hydrology.

In the United States, 64% of the land is dominated by agriculture and silviculture from which the major pollutant is sediment (approximately 1.8 billion metric tons per year) (EPA 1977). The suspended sediments transport toxicants, nutrients, and lower the aesthetic value of the waterways (EPA 1977). Suspended sediments decrease light penetration and photosynthesis, clog gills and filtering systems of aquatic organisms, reduce prey capture, reduce spawning, reduce survival of sensitive species, and carry adsorbed pollutants (Tables 3-3, 3-4, and 3-5). Acute effects of suspended solids are commonly observed at 80,000 mg/L with death at 200,000 mg/L. Recovery is quick at lower exposures (EPA 1977). As the suspended sediments settle, they cover silt-free spawning substrates suffocating embryos and alter the sediment environment. Suspended solids reduce primary productivity and alter temperatures, thus affecting summer stratification. EPA (1976) stated that solids should not reduce photosynthesis by more than 10% of the seasonal average, using the “light-dark” bottle method (APHA 1992). Reduced productivity may then reduce zooplankton populations. Desirable benthic species may be smothered and tolerant species, such as oligochaetes, will increase in numbers. The sediment environment plays a major role in aquatic ecosystem functioning and overlying water quality (Wetzel 1975). These new bedded sediments may possess different chemical, physical, and biological characteristics from pre-impact sediments. So any alteration to the micro-, meio-, and macrobenthic communities, sorption and desorption dynamics of essential and toxic chemical species, and organic matter and nutrient cycling processes, may profoundly influence the aquatic ecosystem (Power and Chapman 1992). As the rate of bedload sediment movement increases and the frequency of occurrence of bedload movement increases, the stress to the system increases.

Dissolved solids concentrations can often be very high in stormwaters and baseflows. The associated dissolved constituents consist primarily of road salts and salts from exposed soils. Though the major cations and anions are nontoxic to most species in relatively high concentrations, stormwaters may exceed threshold levels (EPA 1977) and alter ion ratios which may cause chronic toxicity effects (Ingersoll, *et al*. 1990; 1992). In addition, toxic trace metal-metalloids such as selenium, may be dissolved from natural soil matrices (as dramatically demonstrated in the San Joaquin Valley’s Kesterson Reservoir of California) or dissolved zinc may be discharged from roof runoff components of urban runoff. Long term and repeated exposures result as the dissolved species accumulate in interstitial water, bacteria, macrophytes, phytoplankton, and other food chain components (Burton, *et al*. 1987, EPA 1977) and result in increased mortality, teratogenicity (Hoffman 1988), and other adverse effects (EPA 1977).

Table 3-3. Classification of Suspended and Dissolved Solids and Their Probable Major Impacts on Freshwater Ecosystems (EPA 1977).

	Chemical and	Biochemical and
Physical Effects	Biological Effects*	
Suspended Solids		
Clays, silts, sand	Sedimentation, erosion & abrasion turbidity (light reduction), habitat change	Respiratory interference habitat restriction, light limitation
Natural organic matter	Sedimentation, DO utilization	Food sources, DO effects
Wastewater organic particles	Sedimentation, DO utilization	DO effects, eutrophication,

		nutrient source
Toxicants sorbed to particles	All of the above	Toxicity
<u>Dissolved Solids</u>		
Major inorganic salts	Salinity, buffering, precipitation, element ratios	Nutrient availability, succession, salt effects
Important nutrients		Eutrophication, DO production
Natural organic matter		DO effects and utilization
Wastewater organic matter		DO effects and utilization
Toxicants		Toxicity and effects on DO

Table 3-4. Summary of Suspended Solids Effects on Aquatic Macroinvertebrates (EPA 1977).

Table 3-5. Summary of Suspended Solids Effects on Fish (EPA 1977).

Table 3-5. Summary of Suspended Solids Effects on Fish (EPA 1977) (cont.).

Dissolved Oxygen

Historically, dissolved oxygen has received much attention when investigating biological receiving water effects of pollutant discharges. Therefore, the earliest efforts to evaluate the potential problems caused by urban runoff included investigations of dissolved oxygen conditions in urban receiving waters.

Bacteria respond rapidly (within minutes) in temperate streams and lakes to their surrounding environment. Due to the low level of nutrients normally present, most of the indigenous bacteria are dominant. During a storm event, however, micro- to submicrogram levels of organic nutrients (e.g., carbon, nitrogen, phosphorus, and sulfur containing compounds) suddenly increase by orders of magnitude. Consequently, bacterial reproduction and respiration rates increase dramatically; thus exerting biochemical oxygen demand (BOD). Oxygen depletion problems may occur during the high flow event, but it is likely more serious days later when associated with organic material affecting the sediment oxygen demand (Pitt 1979). BOD_5 levels may exceed 20 mg/L during storm events that may result in anoxia in downstream receiving waters (Schueler 1987). Predicting this problem is complicated by toxicants that may be present and interfere with the BOD test (OWML 1982). Sediment resuspension contributes both BOD and chemical oxygen demand (COD). BOD_5 values were elevated tenfold, 10 to 20 days after a storm event, related to sediment oxygen demand (SOD). Stormwater dissolved oxygen (DO) levels less than 5 mg/L are common (Keefer, *et al*. 1979).

Aquatic macrofauna are cold-blooded and sensitive to temperature changes. In cold water systems, sustained temperatures in excess of 21°C are stressful to resident biota. Many agricultural and urban watersheds contribute to thermal pollution by removing shade canopies over streams and runoff temperatures increase rapidly as they flow over impervious surfaces (Schueler 1987).

Acid precipitation and acid mine drainage are NPS pollution problems in some parts of the United States which are, at times, aggravated by storm events. During the spring in areas where snows have accumulated, rain events intensify the snow melt process. This results in pulses of low pH runoff and snowmelts which may be stressful or lethal to aquatic macrofauna, particularly the sensitive life stages of fish occurring during the spring spawning period.

Keefer, *et al*. (1979) examined the data from 104 water quality monitoring sites near urban areas throughout the country for dissolved oxygen conditions. These stations were selected from more than 1,000 nationwide monitoring stations operated by various federal and state agencies. They conducted analyses of daily dissolved oxygen data for 83 of these sites. About one half of the monitoring stations examined showed a 60 percent or greater, probability of a higher than average dissolved oxygen deficit occurring at times of higher than average streamflow, or on days with rainfall. This result was based upon daily data for entire water years; not all years at any given location exhibited this 60 percent probability condition. They found that the DO levels fell to less than 75 percent saturation at most of the stations that had this 60 percent or greater probability condition. They also found that DO concentrations of less than 5 mg/L were common. Keefer, *et al*. (1979) examined hourly dissolved oxygen data at 22 nationwide sites to find correlations between flows and DO deficit. They found that for periods of steady low flows, the DO fluctuated widely on a daily cycle, ranging from 1 to 7 mg/L. During rain periods, however, the flow increased, of course, but the diurnal cycle of this dissolved oxygen fluctuation disappeared. The minimum DO dropped from 1 to 1.5 mg/L below the minimum values observed during steady flows, and remained constant for periods ranging from 1 to 5 days. They also reported that as the high flow conditions ended, the DO levels resumed diurnal cyclic behavior. About 50 percent of the stations examined in detail on an hour by hour basis would not meet a 5 mg/L DO standard, and about 25 percent of these stations would not even meet a 2.0 mg/L standard for 4-hour averages. The frequency of these violations was estimated to be up to 5 times a year per station.

Ketchum (1978) conducted another study in Indiana that examined dissolved oxygen depletion on a regional basis. Sampling was conducted at nine cities and the project was designed to detect significant dissolved oxygen deficits in streams during periods of rainfall and runoff. The results of this study indicated that wet weather DO levels generally appeared to be similar or higher than those observed during dry weather conditions in the same streams. They found that significant wet weather DO depletions were not observed, and due to the screening nature of the sampling program, more subtle impacts could not be measured.

Heaney, *et al*. (1980), during their review of studies that examined continuous dissolved oxygen (DO) stations downstream from urbanized areas indicated that the worst dissolved oxygen levels occurred after the storms in about one-third of the cases studied. This lowered DO could be due to urban runoff moving downstream, combined sewer overflows and/or resuspension of benthic deposits. Resuspended benthic deposits could have been previously settled urban runoff solids.

Pitt (1979) found that the biochemical oxygen demand of urban runoff, after a 10 to 20 day incubation period, can be more than 5 to 10 times the biochemical oxygen demand of a 1 to 5 day incubation period (Figure 3-29). Therefore, urban runoff effects on dissolved oxygen may occur at times substantially different from the actual storm period and be associated with interaction between sediment and the overlying water column. It is especially important to use acclimated microorganisms for the BOD test seed for stormwater BOD analyses. The standard activated sludge seed may require substantial acclimation periods. Even in natural waters, several day acclimation periods may be needed (see Lalor and Pitt, 1998, P/R *in-situ* test descriptions in Chapter 6).

Figure 3-29. BOD rate curve for stormwater, showing dramatic increase after 10 days of incubation (Pitt 1979).

Temperature

In-stream temperature increases have been noted in many studies as being adversely affected by urbanization. Rainwater flowing across heated pavement can significantly elevate stormwater temperatures. This temperature increase can be very detrimental in steams having sensitive cold-water fisheries. Removal of riparian vegetation can also increase in-stream water temperatures. Higher water temperatures increase the toxicity of ammonia and also affect the survival of pathogens. The temperature increases in urban streams are most important during the hot summer months when the natural stream temperatures may already be nearing critical conditions and when the stream flows are lowest. Pavement is also the hottest at this time and stormwater temperature increases are therefore the highest. Much of the habitat recovery efforts in urban streams focus on restoring an overstory for the streams to provide shading, refuge areas, and bank stability. Wet detention ponds in urban areas have also been shown to cause significant temperature increases. Grass-lined channels, however, provide some relief, compared to rock-lined or asphalt-lined drainage channels. Since temperature is simple to monitor and is a critical stressor for many aquatic organics, it should be included in most monitoring efforts.

Nutrients

In general, urban stormwater is relatively low in organic matter and nutrients and high in toxicants. However, the nutrient levels in stormwaters can periodically be high and produce large mass discharges of nitrogen and phosphorus compounds (e.g., EPA 1977 and 1983, Schueler 1987). Single spring storm events have been shown to contribute 90% of the annual phosphorus input into receiving impoundments (Nix 1976). However, urban and agricultural runoff may contain nutrient concentrations which exceed the normal (pre-development) ranges, and result in adverse responses such as cyanobacterial (blue-green algae) and green algal blooms . Many of the nutrients present in urban runoff are soluble and thus readily assimilated by planktonic organisms (Schueler 1987). Sources include rain, dry deposition, soils, fertilizers, and animal wastes. Impoundment receiving contaminated runoff, with retention times of two weeks or greater, may develop symptoms of eutrophication: Blue-green algal blooms can produce hepato- and neurotoxins implicated in cattle deaths, human liver cancer and allergic responses (Zhang, *et al*. 1991). As algal blooms eventually decompose, bacterial respiration may result in DO sags and anoxia, with associated fish kills.

A large amount of the nutrients enter receiving waters adsorbed to suspended solids (Lin 1972, Middlebooks 1974, Carlile, *et al*. 1974). These fractions will largely end up as bedded sediments which may or may not be subsequently released to overlying waters. The sediment nutrients may stimulate bacterial activity, ammonia production, and rooted macrophyte growth.

Toxicants

Heavy Metals

Stormwater runoff commonly contains elevated levels of metals and metalloids, particularly in urban areas (EPA 1983, Pitt, *et al.* 1995, Schueler 1987). Some of these constituents are very toxic at relatively low concentrations (Table 3-6). The metals of principal concern that often occur in urban runoff are arsenic, cadmium, copper, lead, mercury, and zinc (EPA 1983). Metal bioavailability is reduced in waters of higher hardness (Table 3-6), by sorption to solids, and by stormwater dilution. However, acute and chronic effects have been attributed to stormwater metals (Ray and White 1979, Ellis 1992). The highest metal concentrations are not always associated with the "first flush," but are better correlated with the peak flow period (Heaney 1978). Most metals are bound to street and parking area particulates and subsequently deposited in stream and lake sediments (Pitt, *et al*. 1995). Sediment metal concentrations are dependent on particle size (Wilber and Hunter 1980). Wilber and Hunter (1980) suggest that larger particle sizes are better indicators of urban inputs since they are less affected by scouring. Zinc and copper are often present in runoff as soluble forms (Schueler 1987; Pitt, *et al*. 1995).

Predicting detrimental effects from water or sediment metal concentration or loading data is difficult due to the myriad of processes which control bioavailability and fate. Speciation, availability, and toxicity are affected by pH, redox potential, temperature, hardness, alkalinity, solids, iron and manganese oxyhydroxides, sulfide fractions, and other organic-inorganic chelators. These constituents and conditions are often rapidly changing during a storm event and processes which increase and decrease bioavailability (e.g., loss of sulfide complexes and formation of oxyhydroxide complexes) may occur simultaneously. This makes accurate modeling of toxicity difficult, if not impossible.

Episodic exposures of organisms to stormwaters laden with metals can produce stress and lethality (see also Chapter 6). Ray and White (1979) observed fish death days after exposure and miles downstream after metals were diluted to nondetectable levels. Ellis, *et al.* (1992) showed amphipods bioaccumulated zinc from episodic, *in situ* exposures. Repeated exposures increased their sensitivity and mortality was observed 3 weeks after the storm event.

Toxic Organic Compounds

The types and concentrations of toxic organic compounds that are in stormwaters are driven primarily by land use patterns and automobile activity in the watershed. Most non-pesticide organic compounds originate as washoff from impervious areas in commercial areas having large amounts of automobile startups and/or other high levels of vehicle activities, including vehicle maintenance operations and from heavily traveled roads. The compounds of most interest are the polycyclic aromatic hydrocarbons (PAHs). Other organics include phthalate esters (plasticizers) and aliphatic hydrocarbons. Other compounds frequently detected in residential and agricultural areas are cresol constituents (and other wood preservatives), herbicides, and insecticides. Many of these organic compounds are strongly associated with the particulate fraction of stormwater. Volatile organic compounds (VOCs) are rarely found in urban runoff. While most organics are not detected or are detected at low µg/L concentrations, some are acutely toxic, including freshly applied pesticides and photo-activated PAHs (Skalski 1991, Oris and Giesy 1986). The extent of detrimental impact from these constituents has not been well documented, but likely is significant in some areas.

Environmental Fates of Runoff Toxicants

The fate of runoff toxicants after discharge significantly determines their associated biological effects. If the pollutants are discharged in a soluble form and remain in solution, they may have significant acute toxicity effects on fish, for example. However, if discharged soluble pollutants form insoluble complexes or sorb onto particulates, chronic toxicity effects associated with contaminated sediments are more likely. For many of the metallic and organic toxicants discharged in urban runoff, the particulate fractions are much greater than the soluble fractions (Pitt, *et al*. 1995). Particulate forms of pollutants may remain in suspension, if their settling rates are low and the receiving water is sufficiently turbulent. However, polluted sediments are common in many urban and agricultural streams, indicating significant accumulations of runoff particulate pollutants (Pitt 1995).

Table 3-6. U. S. EPA Trace Metal Criteria for Human Health and Aquatic Life Beneficial Uses

TRACE METAL CONTAMINANT	WATER HARDNESS (mg/L as $CaCO_3$)	HUMAN[1] INGESTION (Food/Drink) (µg/L)	AMBIENT LIFE CRITERIA FOR INTERMITTENT EXPOSURE (µg/L)[2] Threshold[3] Effect	Significant[4] Mortality
Copper	50	-	20	50-90
	100	-	35	90-150
	200	-	80	120-350

Cadmium	50	10	3	7-160
	100	10	6.6	15-350
	300	10	20	45-1070
Lead	50	50	150	350-3200
	100	50	360	820-7500
	200	50	850	1950-17850
Zinc	50	-	380	870-3200
	100	-	680	1550-4500
	200	-	1200	2750-8000
Nickel	-	13.4	-	-

[1] Derived from EPA drinking water criteria.
[2] EPA estimate of toxicity under intermittent, short duration exposure (several hours once every several days).
[3] Concentration causing mortality to the most sensitive individual of the most sensitive species.
[4] Significant mortality shown as a range: 50% mortality in the most sensitive species, and mortality of the most sensitive individual in the species in the 25th percentile of sensitivity.

Tables 3-7 through 3-9 summarize the importance of various environmental processes for the aquatic fates of some runoff heavy metals and organic priority pollutants, as described by Callahan, *et al*. (1979). Photolysis (the breakdown of the compounds in the presence of sunlight) and volatilization (the transfer of the materials from the water into the air as a gas or vapor) are not nearly as important as the other mechanisms for heavy metals. Chemical speciation (the formation of chemical compounds) is very important in determining the solubilities of the specific metals. Sorption (adsorption is the attachment of the material on to the outside of a solid and absorption is the attachment of the material within a solid) is very important for all of the heavy metals shown. Sorption can typically be the controlling mechanism affecting the mobility and the precipitation of most heavy metals. Bioaccumulation (the uptake of the material into organic tissue) can occur for all of the heavy metals shown. Biotransformation (the change of chemical form of the metal by organic processes) is very important for some of the metals, especially mercury, arsenic, and lead. In many cases, the discharge of mercury, arsenic or lead compounds in forms that are unavailable can be accumulated in aquatic sediments. They are then exposed to various benthic organisms that can biotransform the material through metabolization to methylated forms which can be highly toxic and soluble.

Tables 3-8 and 3-9 summarize various environmental fates for some of the toxic organic pollutants found in typical runoff from human-modified watersheds; mainly various phenols, polycyclic aromatic hydrocarbons (PAHs) and phthalate esters. Photolysis may be an important fate process for phenols and PAHs but is probably not important for the phthalate esters. Oxidation or hydrolysis may be important for some phenols. Volatilization may be important for some phenols and PAHs. Sorption is an important fate process for most of the materials, except for phenols. Bioaccumulation, biotransformation and biodegradation are important for many of these organic materials.

Table 3-7. Importance of Environmental Processes on the Aquatic Fates of Selected Urban Runoff Heavy Metals (Callahan, *et al.* 1979)

Environmental Process	Arsenic	Cadmium	Copper	Mercury	Lead	Zinc
Photolysis	Not important	Not important	Not important	May be important in some aquatic environments	Determines the form of lead entering the aquatic system	Not important
Chemical Speciation	Important in determining distribution and mobility (1).	Complexation with organics; most important in polluted waters	Complexation with organics; most important in polluted waters	Conversion to complex species; HgS will precipitate in reducing sediments	Determine which solid phase controls solubility	Complexation predominates in polluted waters
Volatilization	Important when biological	Not important	Not important	Important	Not important	Not important

	activity or highly reducing conditions produce AsH_3 or methly-arsenic					
Sorption	Sorption onto clays, oxides, and organic material important	Sorption onto organic materials, clays, hydrous iron and manganese oxides most important	Can reduce Cu mobility and enrich suspended and bed sediments; sorption onto organics in polluted waters, clay minerals or hydrous iron and manganese oxides	Strongest onto organic material, results in partitioning of mercury into suspended and bed sediments	Adsorption to inorganic solids, organic materials and hydrous iron and manganese oxides control mobility of lead	Strong affinity for hydrous metal oxides, clays and organic matter; adsorption increases with pH
Bioaccumulation	Most important at lower trophic levels; toxicity limits bio-accumulation	Biota strongly bioaccumulate cadmium	Biota strongly bioaccumulate copper	Occurs by many mechanisms, most connected to methylated forms of mercury	Biota strongly bio-accumulates lead	Zinc is strongly bioaccumulated
Biotransformation	Arsenic can be metabolized to organic arsenicals	Not methylized biologically, organic ligands may affect solubility and adsorption	Source Cu complexes may be metabolized; organic ligands are important in sorption and complexation processes	Can be metabolized by bacteria to methyl and dimethly forms which are quite mobile	Biomethylation of lead in sediments can remobilize lead	Not evident; organic ligands of biological origin may affect solubility and adsorption

(1) Conversion of As+3 and As+5 and organic complexation most important.

Table 3-8. Importance of Environmental Processes on the Aquatic Fates of Various Polycyclic Aromatic Hydrocarbons and Phthalate Esters (Callahan, *et al.* 1979)

Environmental Process (1)	*ANTHRACENE*	**Fluoranthene**	**Phenanthrene**	**Diethyl Phthalate (DEP)**	**Di-n-butyl Phthalate (DBP)**	**Bis (2-ethyl-hexyl) Phthalate (DEHP)**	**Butyl Benzyl Phthalate (BBP)**
Photolysis	Dissolved portion may undergo rapid photolysis	Dissolved portion may undergo rapid photolysis	Dissolved portion may undergo rapid photolysis	Not important	Not important	Not important	Not important
Volatilization	May be competitive with adsorption	May be competitive with adsorption	May be competitive with adsorption	Not important	Not important	Not important	Not important

Sorption	Adsorbs onto suspended solids; movement by suspended solids is important transport process	Adsorbs onto suspended solids; movement by suspended solids is important transport process	Adsorbs onto suspended solids; movement by suspended solids is important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process
Bioaccumulation	Short-term process; is readily metabolized	Short-term process; is readily metabolized	Short-term process; is readily metabolized	Variety of organisms accumulate phthalates (lipophilic)	Variety of organisms accumulate phthalates (lipophilic)	Variety of organisms accumulate phthalates (lipophilic)	Variety of organisms accumulate phthalates (lipophilic)
Biotransformation	Readily metabolized by organisms and biodegradation probably ultimate fate mechanisms	Readily metabolized by organisms and biodegradation probably ultimate fate mechanism	Readily metabolized by organisms and biodegradation probably ultimate fate mechanisms	Can be metabolized	Can be metabolized	Can be metabolized	Can be metabolized

(1) Oxidation and hydrolysis are not important fate mechanisms for any of these compounds.

Table 3-9. Importance of Environmental Processes on the Aquatic Fates of Various Phenols and Pyrene (Callahan, *et al.* 1979)

Environmental Process	Phenol	Pentachlorophenol (PCP)	2,4,6-Trichlorophenol	2,4-dimethyl phenol (2,4-xylenol)	Pyrene
Photolysis	Photooxidation may be important degradation process in aerated, clear, surface waters	Reported to occur in natural waters; important near water surface	Reported, but importance is uncertain	May be important degradation process in clear aerated surface waters	Dissolved portion may undergo rapid photolysis
Oxidation	Metal-catalyzed oxidation may be important in aerated surface waters	Not important	Not important	Metal-catalyzed oxidation may be important in aerated surface waters	Not important
Volatilization	Possibility of some phenol passing into the atmosphere	Not important	Not important	Not important	Not as important as adsorption
Sorption	Not important	Sorbed by organic litter in soil and sediments	Potentially important for organic material; not important for clays	Not important	Adsorption onto suspended solids important; movement by suspended solids important
Bioaccumulation	Not important	Bioaccumulates in numerous aquatic organisms	Not important	Not important	Short-term process not significant; metabolized over long-term

Biotransformation	Very significant	Can be metabolized to other phenol forms	Reported in soil and sewage sludge; uncertain for natural surface waters	Inconclusive information	Readily metabolized; biodegradation probably ultimate fate process

Pathogens

Water Environment & Technology (1996) reported that the latest National Water Quality Inventory released by the EPA only showed a slight improvement in the attainment of beneficial uses in the nations waters. Urban runoff was cited as the leading source of problems in estuaries, with nutrients and bacteria as the leading problems. Problems in rivers and lakes were mostly caused by agricultural runoff, with urban runoff the third ranked source for lakes, and the fourth ranked source for rivers. Bacteria, siltation, and nutrients were the leading problems in the nations rivers and lakes.

Pathogens in stormwater are a significant concern potentially affecting human health. The use of indicator bacteria is controversial for stormwater, as well as the assumed time of typical exposure of swimmers to contaminated receiving waters. However, recent epidemiological studies have shown significant health effects associated with stormwater contaminated marine swimming areas. Protozoan pathogens, especially associated with likely sewage-contaminated stormwater, is also of public health concern.

Fecal indicators (i.e., fecal coliforms, fecal streptococcus, *Escherichia coli*, and enterococci) are usually found in elevated concentrations in stormwater runoff, greatly exceeding water quality criteria and standards for primary and secondary contact (MWCOG 1984). This suggests that fecal pathogen levels are also elevated, though significant correlations with fecal coliforms are tenuous (EPA 1986). Die-off of fecal organisms in receiving waters during summer months is relatively rapid, with 99% dying within 24 to 48 hrs (Burton 1985). However, fecal microorganisms also accumulate in sediments where survival is extended for weeks to months (Burton, *et al*. 1987). Current sediment bacteriological analyses being conducted by UAB in local Birmingham area urban lakes have found elevated pore water concentrations (several hundred to several thousand organisms/100 mL) of *E. coli* and enterococci extending to at least 0.1 m in the sediments. Also, when gently disturbed, the water layer over the sediments is also found to significantly increase in microorganism concentrations. Current *in-situ* dieoff studies are also indicating that bacteria sedimentation may be a more important fate mechanism of stormwater bacteria than dieoff.

Good correlations between the incidence of gastroenteritis in swimmers and *E. coli* and enterococci concentrations in water have resulted in new recreational water criteria (EPA 1986). High fecal microorganism concentrations in stormwaters originate from wastes of wildlife, pets, livestock, septic systems, and combined sewer overflows (CSOs). The ecological effects of these inputs of fecal organisms are unknown; however public health is at risk in swimming areas that receive stormwaters.

Urban Bacteria Sources

The Regional Municipality of Ottawa-Carleton (1972) recognized the importance of rooftop, street surface, and field runoff in contributing bacteria contaminants to surface waters in the Ottawa area. Gore and Storrie/Proctor and Redfern (1981) also investigated various urban bacteria sources affecting the Rideau River. They examined dry weather continuous coliform sources, the resuspension of contaminated river bottom sediments, exfiltration from sanitary sewers, and bird feces. These sources were all considered in an attempt to explain the relatively high dry weather coliform bacteria concentrations found in the river. They concluded, however, that stormwater runoff is the most probable source for the wet weather and continuing dry weather bacteria Rideau River concentrations. The slow travel time of the river water usually does not allow the river to recover completely from one rainstorm before another begins.

The Regional Municipality of Ottawa-Carleton (1972) noted the early Ottawa activities in correcting stormwater and sanitary sewage cross-connections. Since that time, many combined sewer overflows have also been eliminated from the Rideau River. Loijens (1981) stated that as a result of sewer separation activities, only one overflow remained active by 1981 (Clegg Street). During river surveys in 1978 and 1979 in the vicinity of this outfall, increased bacteria levels were not found. Gore and Storrie/Proctor and Redfern (1981) stated that there was no evidence that combined sewer overflows are causing the elevated fecal coliform bacteria levels in the river. Environment Canada, (1980) however, stated that high, dry weather bacteria density levels, especially when considering the fecal coliform to fecal strep. ratio, constitutes presumptive evidence of low volume sporadic inputs of sanitary sewage from diverse sources into the downstream Rideau River sectors.

Street surfaces have been identified as potential major sources of urban runoff bacteria. Pitt and Bozeman (1982) found that parking lots, street surfaces, and sidewalks were the major contributors of indicator bacteria in the Coyote Creek watershed in California. Gupta, *et al*. (1981) found high concentrations of fecal coliforms at a highway runoff site in Milwaukee. This site was entirely impervious and located on an elevated bridge deck. The only likely sources of fecal coliforms at this site were atmospheric deposition, bird droppings and possibly feces debris falling from livestock trucks or other vehicles.

Several studies have found that the bacteria in stormwater in residential and light commercial areas were from predominantly nonhuman origins. Geldreich and Kenner (1969) stated that the fecal coliforms in stormwater are from dogs, cats, and rodents in city areas, and from farm animals and wildlife in rural areas. Qureshi and Dutka, (1979) found that there may be an initial flush of animal feces when runoff first develops. However, the most important bacteria source for runoff is the feces bacteria that have been distributed generally in the soils and on the surfaces of the drainage area. The most important source, however, may be feces bacteria that are distributed in the soil and not the fresh feces washing off the impervious surfaces.

Some studies have investigated vegetation sources of coliform bacteria. For example, Geldreich (1965) found that the washoff of bacteria from vegetation does not contribute significant bacteria to the runoff. They also found that most of the bacteria on vegetation is of insect origin. Geldreich, *et al*. (1980) found that recreation activities in water bodies also increase the fecal coliform and fecal strep. concentrations. These organisms of intestinal origin will concentrate in areas near the shore or in areas of stratification. Fennell, *et al.* (1974) found that open dumps containing domestic refuse can be a reservoir of *Salmonella* bacteria that can be spread to nearby water bodies by foraging animals and birds.

When a drainage basin has much of its surface paved, the urban runoff bacteria concentrations can be expected to peak near the beginning of the rainfall event and then decrease as the event continues. Initial high levels of bacteria may be associated with direct flushing of feces from paved surfaces. These feces are from dogs defecating on parking lots and street areas and from birds roosting on rooftops. When a drainage area has a lot of landscaped areas or open land, relatively high bacteria concentrations in the urban runoff may occur throughout the rain event associated with continuous erosion of contaminated soils.

Fecal Coliform to Fecal Strep. Bacteria Ratios

Geldreich (1965) found that the ratio of fecal coliform to fecal strep. bacteria concentrations may be indicative of the probable fecal source. In fresh human fecal material and domestic wastes, he found that the fecal coliform densities were more than four times the fecal strep. densities. However, this ratio for livestock, poultry, dogs, cats, and rodents was found to be less than 0.6. These ratios must be applied carefully because of the effects of travel time and various chemical changes (especially pH) on the dieoff rates of the component bacteria. This can result in the ratio changing, as the fecal coliform organisms tend to die faster than the fecal strep. Bacteria. As a generality, he stated that fecal coliform to fecal strep. ratios greater than four indicate that the bacteria pollution is from domestic wastes, which are composed mostly of human fecal material, laundry wastes, and food refuse. If the ratio is less than 0.6, the bacteria is probably from livestock or poultry in agricultural areas or from stormwater runoff in urban areas. He found that agricultural and stormwater runoff can be differentiated by studying the types of fecal strep. bacteria found in the water samples. Geldreich and Kenner (1969) further stressed the importance of carefully using this ratio. They stressed that samples must be taken at the wastewater outfalls. At these locations, domestic waste, meat packing wastes, stormwater discharges, and feedlot drainage contain large numbers of fecal organisms recently discharged from warm blooded animals. Once these organisms are diffused into the receiving stream, however, water temperature, organic nutrients, toxic metals, and adverse pH values may alter the relationship between the indicator organisms. This ratio should only be applied within 24 hours following the discharge of the bacteria.

Feachem (1975) examined how these ratios could be used with bacteria observations taken over a period of time. Because the fecal coliform and fecal strep. bacteria dieoff rates are not the same, the ratio gradually changes with time. He found that bacteria is predominantly from human sources if the FC/FS ratios are initially high (greater than four) and then decrease with time. Non-human bacteria sources would result in initially low fecal coliform to fecal strep. ratios (less than 0.7) which then rise with time.

Pitt (1983) examined the fecal coliform to fecal strep. bacteria population ratios observed in the Rideau River study area in Ottawa, as shown on Table 3-10. These ratios were divided into groups corresponding to source area samples, Rideau River water samples, and water samples collected at the swimming beaches further downstream. The source area sheet-flow samples contained the most recent pollution, while the river segment and beach samples contained “older” bacteria. The initial source area samples all had ratios of less than 0.7. However, the river averages ranged from 0.5 to 1.2 and the beach samples (which may be “older” than the river samples) ranged from 1.7 to 2.8. These ratios are seen to start with values less than 0.7 and increase with time. Based on Feachem’s (1975) work, this would indicate that the major bacteria sources in the Rideau River are from non-human sources. Periodic high bacteria ratios in the river and at the beaches could be caused by the greater dieoff ratio of fecal strep. as compared to fecal coliform. The observed periodic high Rideau River FC/FS ratios (which can be greater than four) may therefore be from old, non-human fecal discharges and not from fresh human fecal discharges.

Table 3-10. Fecal Coliform to Fecal Strep. Bacteria Population Ratios in Study Area (Pitt 1983)

Source Areas	FC/FS ratio
Rooftop runoff	0.5
Vacant land sheetflow	0.3
Parking lot sheetflow	0.2
Gutter flows	0.2
Average of source area values	0.3
Rideau River Segment	
A	1.2
B	0.6
C	0.5
D	0.5
E	1.0
Average of river segment values	0.7
River Swimming Beaches	
Strathcona	2.8
Brantwood	2.3
Brighton	2.1
Mooney's Bay	1.7
Average of swimming beach values	2.2

Human Health Effects of Stormwater

There are several mechanisms where stormwater exposure can cause potential human health problems. These include exposure to stormwater contaminants at swimming areas affected by stormwater discharges, drinking water supplies contaminated by stormwater discharges, and the consumption of fish and shellfish that have been contaminated by stormwater pollutants. Understanding the risks associated with these exposure mechanisms is difficult and not very clear. Receiving waters where human uses are evident are usually very large and the receiving waters are affected by many sanitary sewage and industrial point discharges, along with upstream agricultural nonpoint discharges, in addition to the local stormwater discharges. In receiving waters only having stormwater discharges, it is well known that inappropriate sanitary and other wastewaters are also discharging through the storm drainage system. These "interferences" make it especially difficult to identify specific cause and effect relationships associated with stormwater discharges alone, in contrast to the many receiving water studies that have investigated ecological problems that can more easily study streams affected by stormwater alone. Therefore, much of the human risk assessment associated with stormwater exposure must use theoretical evaluations relying on stormwater characteristics and laboratory studies in lieu of actual population studies. However, some site investigations, especially related to swimming beach problems associated with nearby stormwater discharges, have been conducted and are summarized (from Lalor and Pitt 1998) in the following discussion.

Contact recreation in pathogen contaminated waters has been studied at many locations. The sources of the pathogens are typically assumed to be sanitary sewage effluent, or periodic industrial discharges from certain food preparation industries (especially meat packing and fish and shellfish processing). However, several studies have investigated pathogen problems associated with stormwater discharges. It has generally been assumed that the source of pathogens in stormwater are from inappropriate sanitary connections. However, stormwater unaffected by these inappropriate sources still contains high counts of pathogens that are also found in surface runoff samples from many urban surfaces. Needless-to-say, sewage contamination of urban streams is an important issue that needs attention during a receiving water investigation.

Inappropriate Sanitary Sewage Discharges into Urban Streams

Urban stormwater runoff includes waters from many other sources that find their way into storm drainage systems, besides from precipitation. There are cases where pollutant levels in storm drainage are much higher than they would otherwise be because of excessive amounts of contaminants that are introduced into the storm drainage system by various non-stormwater discharges. Additionally, baseflows (during dry weather) are also common in storm drainage systems. Dry-weather flows and wet-weather flows have been monitored during numerous urban runoff studies. These studies have found that discharges observed at outfalls during dry weather were significantly different from wet-weather discharges and may account for the majority of the annual discharges for some pollutants of concern from the storm drainage system.

In many cases, sanitary sewage was an important component (although not necessarily the only component) of the dry weather discharges from the storm drainage systems. From a human health perspective (associated with pathogens), it may not require much raw or poorly treated sewage to cause a receiving water problem. However, at low discharge rates, the DO receiving water levels may be minimally affected. The effects these discharges have on the receiving waters is therefore highly dependent on many site specific factors, including frequency and quantity of sewage discharges and the creek flows. In many urban areas, the receiving waters are small creeks in completely developed watersheds. These creeks are the most at risk from these discharges as dry base flows may be predominately dry weather flows from the drainage systems. In Tokyo (Fujita 1998), for example, numerous instances were found where correcting inappropriate sanitary sewage discharges resulted in the urban streams losing all of their flow. In cities that are adjacent to large receiving waters, these discharges likely have little impact (such as DO impacts from Nashville CSO discharges on the Cumberland River, as studied by Cardozo, *et al.* 1994). The presence of pathogens from raw, or poorly treated sewage, in urban streams, however, obviously presents a potentially serious public health threat. Even if the receiving waters are not designated as water contact recreation, children are often seen playing in small city streams.

There have been a few epidemiology studies recently published describing the increased health risks associated with contaminated dry weather flows affecting public swimming beaches. The following discussion presents an overview of the development of water quality criteria for water contact recreation, plus the results of a recent epidemiological study that specifically examined human health problems associated with swimming in water affected by stormwater. In most cases, the levels of indicator organisms and pathogens causing increased illness were well within the range found in urban streams.

Runoff Pathogens and Their Sanitary Significance

The occurrence of *Salmonella* biotypes is typically low and their reported density is less than one organism/100mL in stormwater. *Pseudomonas aeruginosa* are frequently encountered at densities greater than ten organisms/100mL, but only after rains. The observed ranges of concentrations and percent isolations of bacterial biotypes vary significantly from site to site and at the same location for different times. Many potentially pathogenic bacteria biotypes may be present in urban runoff. Because of the low probability of ingestion of urban runoff, many of the potential human diseases associated with these biotypes are not likely to occur. The pathogenic organisms of most concern in urban runoff are usually associated with skin infections and body contact. The most important biotype causing skin infections would be *Pseudomonas aeruginosa*. This biotype has been detected frequently in most urban runoff studies in concentrations that may cause potential infections. However, there is little information associating the cause and effect of increased *Pseudomonas* concentrations with increased infections. Shigella may be present in urban runoff and receiving waters. This pathogen, when ingested in low numbers, can cause dysentery.

Salmonella. *Salmonella* has been reported in some, but not all, urban stormwaters. Qureshi and Dutka (1979) frequently detected *Salmonella* in southern Ontario stormwaters. They did not find any predictable patterns of *Salmonella* isolations as they were found throughout the various sampling periods. Olivieri, *et al.* (1977a) found *Salmonella* frequently in Baltimore runoff, but at relatively low concentrations. Typical concentrations were from five to 300 *Salmonella* organisms/ten liters. The concentrations of *Salmonella* were about ten times higher in the stormwater samples than in the urban stream receiving the runoff. They also did not find any marked seasonal variations in *Salmonella* concentrations. Almost all of the stormwater samples that had fecal coliform concentrations greater that 2000 organisms/100 mL had detectable *Salmonella* concentrations, while about 27 percent of the samples having fecal coliform concentrations less than 200 organisms/100 mL had detectable *Salmonella*.

Quite a few urban runoff studies have not detect *Salmonella*. Schillinger and Stuart (1978) found that *Salmonella* isolations were not common in a Montana subdivision runoff study and that the isolations did not correlate well with fecal coliform concentrations. Environment Canada (1980) stated that *Salmonella* were virtually absent from Ottawa storm drainage samples in 1979. They concluded that *Salmonella* are seldom present in significant numbers in Ottawa urban runoff. The types of *Salmonella* found in southern Ontario were *S. thompson* and *S. typhimurium var copenhagen* (Qureshi and Dutka 1979).

Olivieri, *et al*. (1977b) stated that the primary human enteric disease producing *Salmonella* biotypes associated with the ingestion of water include *S. typhi* (typhoid fever), *S. paratyphi* (paratyphoid fever), and *Salmonella* species (salmonellosis). These biotypes are all rare except for *Salmonella*. The dose of *Salmonella* required to produce an infection is quite large (approximately 105 organisms). The salmonellosis health hazard associated with water contact in urban streams is believed to be small because of this relatively large infective dose. If two liters of stormwater having typical *Salmonella* concentrations (ten *Salmonella* organisms per/ten liters) is ingested, less than 0.001 of the required infective dose would be ingested. If a worse case *Salmonella* stormwater concentration of 10,000 organisms/ten liters occurred, the ingestion of 20 liters of stormwater would be necessary for an infective dose. They stated that the low concentrations of *Salmonella*, coupled with the unlikely event of consuming enough stormwater, make the *Salmonella* health hazard associated with urban runoff small.

Staphylococci. *Staphylococcus aureus* is an important human pathogen as it can cause boils, carbuncles, abscesses, and impetigo on skin on contact. Olivieri, *et al.* (1977b) stated that the typical concentrations of Staphylococci are not very high in urban streams. They also noted that there was little information available relating the degree of risk of staph. infections with water concentrations. They concluded that *Staph. aureus* appears to be the most potentially hazardous pathogen associated with urban runoff, but there is no evidence available that skin, eye, or ear infections can be caused by the presence of this organism in recreational waters. They concluded that there is little reason for extensive public health concern over recreational waters receiving urban storm runoff containing staph. organisms.

Shigella. Olivieri, *et al.* (1977b) stated that there is circumstantial evidence that Shigella is present in urban runoff and receiving waters and could present a significant health hazard. Shigella species causing bacillary dysentery are one of the primary human enteric disease producing bacteria agents present in water. The infective dose of Shigella necessary to cause dysentery is quite low (ten to 100 organisms). Because of this low required infective dose and the assumed presence of Shigella in urban waters, it may be a significant health hazard associated with urban runoff.

Streptococcus. *Streptococcus faecalis* and atypical *S. faecalis* are of limited sanitary significance (Geldreich 1976). Streptococcus determinations on urban runoff are most useful for identifying the presence of *S. bovis* and *S. equinus* that are specific indicators of non-human, warm blooded animal pollution. However, it is difficult to interpret fecal strep. data when their concentrations are lower than 100 organisms/100 mL because of the ubiquitious occurrence of *S. faecalis var. liquifaciens*. This biotype is generally the predominant strep. biotype occurring at low fecal strep. concentrations.

Pseudomonas aeruginosa. *Pseudomonas* is reported to be the most abundant pathogenic bacteria organism in urban runoff and streams (Olivieri, *et al.*(1977b). This pathogen is associated with eye and ear infections and is resistant to antibiotics. They also stated that past studies have failed to show any relationships between *P. aeruginosa* concentrations in bathing waters and ear infections. However, *Pseudomonas* concentrations in urban runoff are at significantly greater concentrations (about 100 items) than the values associated with past bathing beach studies. Cabelli, *et al*. (1976) stated that *Pseudomonas aeruginosa* is indigenous in about 15 percent of the human population. Swimmer's ear or other *Pseudomonas* infections may, therefore, be caused by trauma to the ear canals associated with swimming and diving, and not exposure to *Pseudomonas* in the bathing water.

Environment Canada (1980) stated that there is preliminary evidence of the direct relationship between very low levels of *Pseudomonas aeruginosa* and an increase in incidents of ear infections in swimmers. They stated that a control level for this *Pseudomonas* biotype of between 23 and 30 organisms/100 mL is being considered. Cabelli, *et al.* (1976) stated that *P. aeruginosa* densities greater than ten organisms/100 mL were frequently associated with fecal coliform levels considerably less than 200 organisms/100 mL. *P. aeruginosa* densities were sometimes very low when the fecal coliform levels were greater than 200 organisms/100 mL. An average estimated *P. aeruginosa* density associated with a fecal coliform concentration of 200 organisms/100 mL is about 12/100 mL. They further stated that *P. aeruginosa* by itself cannot be used as a basis for water standards for the prevention of enteric diseases during recreational uses of surface waters. The determinations of this biotype should be used in conjunction with fecal coliform or other indicator organism concentrations for a specific location. They recommended that bathing beaches that are subject to urban runoff pollution be temporarily closed until the *P. aeruginosa* concentrations return to a baseline concentration.

Campylobacter. Koenraad, *et al*. (1997) investigated the contamination of surface waters by *Campylobacter* and its associated human health risks. They reported that campylobacteriosis is one the most frequently occurring acute gastroenteritis diseases in humans. Typical investigations have focused on the consumption of poultry, raw milk, and untreated water as the major sources of this bacterial illness. Koenraad, *et al*. (1997) found that human exposures to *Campylobacter* contaminated surface waters is likely a more important risk factor than previously considered. In fact, they felt that *Campylobacter* infections may be more common than *Salmonella* infections. The incidence of campylobacteriosis due to exposure to contaminated recreational waters has been estimated to be between 1.2 to 170 per 100,000 individuals. The natural habitat of *Campylobacter* is the intestinal tract of warm-blooded animals (including poultry, pigs, cattle, gulls, geese, pigeons, magpies, rodents, shellfish, and even flies). It does not seem to multiply outside of its host, but it can survive fairly well in aquatic environments. It can remain culturable and infective for more than 2 months under ideal environmental conditions. Besides runoff, treated wastewater effluent is also a major likely source of *Campylobacter* in surface waters. Sanitary wastewater may contain up to 50,000 MPN of *Campylobacter* per 100 mL, with 90 to 99% reductions occurring during typical wastewater treatment.

Cryptosporidium, Giardia, and Pfiesteria. Protozoa became an important public issue with the 1993 *Cryptosporidium*-caused disease outbreak in Milwaukee when about 400,000 people become ill from drinking contaminated water. Mac Kenzie, *et al*. (1994) prepared an overview of the outbreak, describing the investigation on the causes of the illness and the number of people affected. They point out that *Cryptosporidium*-caused disease in humans was first documented in 1976, but had received little attention and no routine monitoring. *Cryptosporidium* now is being monitored routinely at many areas and is the subject of much research concerning its sources and pathways. At the time of the Milwaukee outbreak, both of the city's water treatment plants (using water from Lake Michigan) were operating within acceptable limits, based on required monitoring. However, at one of the plants (which delivered water to most of the infected people), the treated water experienced a large increase in turbidity (from about 0.3 NTU to about 1.5 NTU) at the time of the outbreak that was not being well monitored (the continuous monitoring equipment was not functioning, and values were only obtained every 8 hours). More than half of the residents receiving water from this plant became ill. The plant had recently changed its coagulant from polyaluminum chloride to alum and equipment to assist in determining the correct chemical dosages was not being used. The finished water had apparently relatively high levels of cryptosporidium because some individuals became ill after only drinking less than 1 L of water.

Cryptosporidium oocysts have often been found in untreated surface waters, and it was thought that *Cryptosporidium* oocysts entered the water treatment supply before the increase in turbidity was apparent. Mac Kenzie, *et al.* (1994) point out that monitoring in the United Kingdom has uncovered sudden, irregular, community-wide increases in cryptosporidiosis that were likely caused by waterborne transmission. They also stated that the source of the *Cryptosporidium* oocysts was speculative, but could have included cattle feces contamination in the Milwaukee and Menomonee Rivers, slaughterhouse wastes, and human sewage. The rivers were also swelled by high spring rains and snowmelt runoff that may have aided the transport of upstream *Cryptosporidium* oocysts into the lake near the water intakes.

The *Journal of the American Water Works Association* has published numerous articles on protozoa contamination of drinking water supplies. Crockett and Haas (1997) describe a watershed investigation to identify sources of *Giardia* and *Cryptosporidium* in the Philadelphia watershed. They describe the difficulties associated with monitoring *Cryptosporidium* and *Giardia* in surface waters because of low analytical recoveries and the cost of analyses. Large variations in observed protozoa concentrations made it difficult to identify major sources during the preliminary stages of their investigations. They do expect that wastewater treatment plant discharges are a major local source, although animals (especially calves and lambs) are likely significant contributors. Combined sewer overflows had *Giardia* levels similar to raw sewage, but the CSOs were much less than the raw sewage for *Cryptosporidium*. LeChevallier, *et al*. (1997) investigated *Giardia* and *Cryptosporidium* in open reservoirs storing finished drinking water. This gave them an opportunity to observe small increases in oocyst concentrations associated from nonpoint sources of contamination from the highly controlled surrounding area. They observed significantly larger oocyst concentrations at the effluent (median values of 6.0 *Giardia*/100 L and 14 *Cryptosporidium*/100 L) in the reservoirs than in the influents (median values of 1.6 *Giardia*/100 L and 1.0 *Cryptosporidium*/100 L). No human wastes could influence any of the tested reservoirs and the increases were therefore likely caused by wastes from indigenous animals or birds, either directly contaminating the water, or through runoff from the adjacent wooded areas.

A Management Training Audioconference Seminar on *Cryptosporidium* and Water (MTA 1997) was broadcast in May of 1997 to familiarize state and local agencies about possible *Cryptosporidium* problems that may be evident as a result of the EPA's Information Collection Rule which began in July of 1997. This regulation requires all communities serving more than 100,000 people to monitor their source water for *Cryptosporidium* oocysts. If the source water has more than 10 *Cryptosporidium* oocysts per liter, then the finished water must also be monitored. It is likely that many source waters will be found to be affected by cryptosporidium. They reviewed one study that found the percentage of positive samples of *Cryptosporidium* in lakes, rivers, and springs was about 50 to 60% and about 5% in wells. In contrast, the percentage of samples testing positive for *Giardia* was about 10 to 20% in lakes and rivers, and very low in springs and wells.

Special human health concerns have also been recently expressed about *Pfiesteria piscicida*, a marine dinoflagellate that apparently is associated with coastal eutrophication caused by runoff nutrients (Maguire and Walker 1997). Dramatic blooms and resulting fish kills have been associated with increased nutrient loading from manure-laden runoff from large livestock feedlot operations. This organism has gathered much attention in the popular press, usually called the "cell from hell" (Zimmerman 1998). It has been implicated as causing symptoms of nausea, fatigue, memory loss, and skin infections in south Atlantic coastal bay watermen. *Pfiesteria* and *Pfiesteria*-like organisms have also been implicated as the primary cause of many major fish kills and fish disease events in Virginia, Maryland, North Carolina, and Delaware. In August of 1997, hundreds of dead and dying fish were found in the Pocomoke River, near Shelltown, Maryland, in the Chesapeake Bay, prompting the closure of a portion of the river. Subsequent fish kills and confirmed occurrences of *Pfiesteria* led to further closures of the Manokin and Chicamacomico Rivers. The Maryland Department of Health and Mental Hygiene also presented preliminary evidence that adverse public health effects could results from exposure to the toxins released by *Pfiesteria* and *Pfiesteria*-like organisms. The increasing numbers of fish kills of Atlantic menhaden (an oily, non-game fish) motivated Maryland's governor to appoint a Citizens *Pfiesteria* Action Commission. The Commission conveyed a forum of noted scientists to examine the existing information on *Pfiesteria.* The results of the State of Maryland's *Pfiesteria* monitoring program area available on the Maryland Department of Natural Resources' website:
http://www.dnr.state.md.us/pfiesteria/

Pfiesteria has a complex life cycle, including at lease 24 flagellated, amoeboid, and encysted stages. Only a few of these stages appears to be toxic, but their complex nature makes them difficult to identify by nonexperts (Maguire and Walker 1997). *Pfiesteria* spends much of its life span in a nontoxic predatory form, feeding on bacteria and algae, or as encysted dormant cells in muddy sediment. Large schools of oily fish (such as the Atlantic menhaden) trigger the encysted cells to emerge and excrete toxins. These toxins make the fish lethargic, so they remain in the area where the toxins attack the fish skin, causing open sores to develop. The *Pfiesteria* then feed on the sloughing fish tissue. Unfortunately, people working in the water during these toxin releases may also be affected (Zimmerman 1998).

Researchers suggest that excessive nutrients (causing eutrophication) increase the algae and other organic matter that the *Pfiesteria* and Atlantic menhaden use for food. The increased concentrations of *Pfiesteria* above natural background levels increase the likelihood of toxic problems. Maguire and Walker (1997) state that other factors apparently are also involved, including stream hydraulics, water temperature, and salinity. They feel that *Pfiesteria* is only one example of the increasing threats affecting coastal ecosystems that are experiencing increased nutrient levels. Most of the resulting algal blooms only present nuisance conditions, but a small number can result in human health problems (mostly as shellfish poisonings). The increased nutrient discharges are mostly associated with agricultural operations, especially animal wastes from large poultry and swine operations. In the Pocomoke River watershed, the Maryland Department of Natural Resources estimates that about 80% of the phosphorus and 75% of the nitrogen load is from agricultural sources. Urban runoff may also be a causative factor of eutrophication in coastal communities, especially those having small enclosed coastal lagoons or embayments, or in rapidly growing urban areas. Zimmerman (1998) points out that the Chesapeake Bay area is one of the country's most rapidly growing areas, with the population expected to increase by 12 percent by the year 2010.

Viruses. It is believed that approximately half of all waterborne diseases are of viral origin. Unfortunately, it is very difficult and time consuming to identify viruses from either environmental samples or sick individuals. When the EPA conducted its extensive epidemiological investigations of freshwater and marine swimming beaches (discussed above) in the 1980's, two viruses common to human gastrointestinal tracts (coliphage and enterovirus) were evaluated as potential pathogen indicators. These two indicators did not show good correlations between their presence and the incidence of gastroenteritis. Viruses tend to survive for slightly longer periods in natural waters than do gram negative bacteria. It is believed that the high correlation observed between gastroenteritis and the presence of enterococci may be because the gram positive enterococci's longer survival more closely mimics viral survival. Therefore, enterococci may serve as a good recreational water indicator for the presence of viral pathogens.

Receiving Water Effect Summary

Recent studies (discussed above) have combined chemical-physical characterizations of water and sediment, with biosurveys and laboratory/*in situ* toxicity surveys (low and high flow) to effectively characterized major water column and sediment stressors (Burton and Rowland 1999; Burton, *et al.* 1998; Dyer and White; Burton and Moore 1999). Suspended solids, ammonia, sediments, temperature, PAHs, sediment, and/or stormwater runoff were observed to be primary stressors in these test systems. These primary stressors could not have been identified without low and high flow and sediment quality assessments both in the laboratory and field. It is apparent that in order to determine the role of chemicals as stressors in the receiving waters, the role of other stressors (both natural and anthropogenic) must be assessed (see also Chapters 6 and 8).

Johnson, *et al.* (1996) and Herricks, *et al.* (1996a, 1996b) describe a structured tier testing protocol to assess both short-term and long-term wet weather discharge toxicity. The protocol recognizes that the test systems must be appropriate to the time-scale of exposure during the discharge. Therefore, three time-scale protocols were developed, for intra-event, event, and long-term exposures.

There is a natural tendency in the popular "weight-of-evidence" or "sediment quality triad"- type approaches to look for "validation" of one assessment tool with another (see also Chapters X and 8). For example, matching a toxic response in a WET test with that of an impaired community gives a greater weight of evidence. This does not, however, necessarily "validate" the results (or invalidate if there are differences) (Chapman 1995b). Natural temporal changes in aquatic populations at different sites within a study system need not be the same (Power, *et al.* 1988; Resh 1988; Underwood 1993), therefore, predictions of effect or no-effect from WET testing of reference sites may be in error. Each monitoring tool (i.e., chemical, physical and indigenous biota characterizations, laboratory and field toxicity and bioaccumulation) provides unique and often essential information (Burton 1995b; Chapman, *et al.* 1992, Burton, *et al*. 1996; Burton and Baird 2000). If responses of each of the biological tools disagree, it is likely due to species differences or a differing stressor exposure dynamics/interactions. These critical exposures issues can be characterized through a systematic process of separating stressors and their respective dynamics into low and high flow and sediment compartments using both laboratory and field exposures. Then a more efficient and focused assessment can identify critical stressors and determine their ecological significance with less uncertainty than the more commonly used approaches. The chronic degradation potential of complex ecosystems receiving multiple stressors cannot be adequately evaluated without a comprehensive assessment that characterizes water, sediment and biological dynamics and their interactions.

Because most sites have multiple stressors (physical, chemical, and biological), it is essential that the relative contribution of these stressors be defined to effectively design corrective measures. The integrated laboratory and field approach rigorously defines the exposures of organisms (media of exposure and contaminant concentration), separating it into overlying water, surficial sediment, historical sediment, and interstitial water. The degree of contaminant-associated toxicity can best be assessed using a combination of laboratory and field screening methods which separate stressors (*i.e*., a Stressor Identification Evaluation (SIE) approach) (Burton, *et al*. 1996), into different, major stressor categories, including: metals, nonpolar organics, photo-induced toxicity from PAHs, ammonia, suspended solids, predators, dissolved oxygen, and flow. There is much research to be done to refine these approaches, but the tools are there already to make ecologically relevant assessments of aquatic ecosystem contamination with reasonable certainty.

The effects of urban runoff on receiving water aquatic organisms or other beneficial uses is also very site specific. Different land development practices may create substantially different runoff flows. Different rain patterns cause different particulate washoff, transport and dilution conditions. Local attitudes also define specific beneficial uses and desired controls. There are also a wide variety of water types receiving urban and agricultural runoff, and these waters all have watersheds that are urbanized to various degrees. Therefore, it is not surprising that runoff effects, though generally dramatic, are also quite variable and site specific.

Previous attempts to identify runoff problems using existing data have not been conclusive because of differences in sampling procedures and the common practice of pooling data from various sites, or conditions. It is therefore necessary to carefully design comprehensive, long-term studies to investigate runoff problems on a site-specific basis. Sediment transport, deposition, and chemistry play key roles in receiving waters and need additional research. Receiving water aquatic biological conditions, especially compared to unaffected receiving waters, should be studied in preference to laboratory bioassays.

These specific studies need to examine beneficial uses directly, and not rely on published water quality criteria and water column measurements alone. Published criteria are usually not applicable to urban runoff because of the slug nature of runoff and the unique chemical speciation of its components.

The long-term aquatic life effects of runoff are probably more important than short-term effects associated with specific events. The long-term effects are probably related to the deposition and accumulation of toxic sediments, or the inability of the aquatic organisms to adjust to repeated exposures to high concentrations of toxic materials or high flow rates.

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III. Water Quality Conditions in the Cahaba River and Likely Pollutant Sources

Excerpted from a report by Robert Pitt to the Alabama Department of Environmental Management, April 2000.

Introduction

This report summaries preliminary assessments of the historical water quality conditions, as related to applicable water quality standards, found in the Cahaba River. This assessment focuses on pollutants (especially toxicants) observed along the upper and middle reaches of the Cahaba River. Additional work is needed to evaluate water quality conditions in the other portions of the river, to better estimate the magnitude of some of the likely pollutant sources of the problem pollutants, and to update the evaluation using more recent data. This report was originally prepared for the Alabama Department of Environmental Management (ADEM) Commission's Cahaba River Work Group in 1990 and for Torchmark Corp. in 1994.

Most of the water quality data were obtained from the U.S. EPA's STORET computer system for 1970 through 1990 directly from the EPA's Atlanta office. The majority of the STORET data was submitted by the Alabama Department of Environmental Management (ADEM), the U.S. Geological Survey (USGS), the Birmingham Water Works Board (BWWB), and the Geological Survey of Alabama. Additional data that were not submitted to STORET were obtained directly from the Birmingham Water Works Board. A great deal of time was spent in conducting quality assurance evaluations on the data to eliminate obvious erroneous data before statistical analyses and to locate and plot the sampling locations on 7 1/2 minute USGS quadrangle maps.

Much of the analyses in this report focuses on water quality conditions in the upper Cahaba River watershed above the Highway 280 crossing. This portion of the watershed is the major water supply for the Birmingham, AL, area and is under heavy development pressure. However, no data pertaining to the Little Cahaba River is included. About 1500 samples were included in the initial data collection for the upper portion of the watershed. However, many of the sample data obtained were from areas outside of the area of interest, had apparent mistakes that were not capable of being clarified, or were duplicates (especially between STORET and BWWB). It took a great deal of time to do an adequate quality assurance review of the data. The final data set had 857 unique samples collected between March 11, 1970 and July 25, 1990. Most of samples were obtained in the mid to late 1980s. Only the BWWB pump station location had samples as old as 1970. Sampling from the other locations did not start until about 1975. Also, many of the sampling locations were only represented by a few samples obtained over a short sampling period. However, representative samples for a relatively long period were available for a number of key locations. The most common measurements were performed on about 500 samples, while a few of the parameters of interest were only available for about 150 samples.

More than 40 sampling locations were represented along the complete length of the river in the area upstream of the Highway 280 crossing in the Cahaba River watershed. Probably the most important collection of data was obtained from the BWWB pump station. The pump station data was influenced by flows from the Little Cahaba River, even though no monitoring data from the Little Cahaba River was directly included in this data review. A few of the sampling locations had most of the data, including the BWWB pump station (102 samples), at 16 miles upstream of the pump station (77 samples), and at 28 miles upstream of the pump station (194 samples). Six other locations had between 25 and 40 samples and 14 other locations had between 10 and 25 samples. The remaining locations had fewer than 10 samples each.

Table 1 lists the sample location codes, as shown on the maps and data files, their source, and location along the river, for sampling locations that had data that was used in these analyses. A number of other sampling locations were also identified, but had no useful data for these analyses (usually only infrequent flow information). The river miles is the distance upstream from the BWWB pump station, and if the sample is on a tributary, the distance upstream along the tributary from the Cahaba River is also shown.

Many of the tributaries were not named on the USGS maps and were therefore given arbitrary numbers, as shown. These sampling locations were geographically divided into five areas, each having enough samples for statistical comparisons:

- Area 1 was the BWWB pump station, having 102 samples.
- Area 2 was above the pump station to 14 miles upstream, having 136 samples. Also contains unnamed tributaries 100, 800, and 900.
- Area 3 was from 14 to 20 miles upstream, having 109 samples. Also contains unnamed tributary 1000 and Stinking Creek.
- Area 4 was 20 to 23.5 miles upstream, having 82 samples. Also contains unnamed tributary 200 and Big Black Creek, a major tributary.
- Area 5 was above 23.5 miles upstream, having 430 samples. Also contains No. and So. Forks of Little Cahaba Creek and Pinchgut Creek.

Area 5 was also subdivided to examine the uppermost Cahaba River data separately from both forks of the Little Cahaba Creek and Pinchgut Creek.

Further analyses were conducted using EPA's STORET data for two locations further downstream along the Cahaba River (West Blocton and Centreville). Most of these data were obtained by the State of Alabama (especially ADEM) and submitted to the EPA. These data were compared to the water quality criteria associated with the protection of fish and wildlife and also for the protection of human health associated with the consumption of fish. These downstream locations had long-term data collected during the same 1980 – 1990 time frame as the upper Cahaba River data, and were used to examine the consistency of the problems observed upstream, and to roughly calculate allowable discharges to the river.

Water Quality Criteria

The EPA (1986) has published guidelines for how their criteria are to be applied: "criteria present scientific data and guidance of the environmental effects of pollutants which can be useful to derive regulatory requirements based on consideration of water quality impacts." Being criteria, they are not legal standards but are indicative of problems that may occur if they are exceeded. However, many states, including Alabama, have adopted many of the EPA criteria as enforceable standards. In most cases, the EPA's criteria are contained in the Alabama standards. Notable exceptions are the lack of a nitrate standard for drinking water supplies and an arsenic standard to protect consumers of fish in Alabama.

Table 1. Sampling Locations for Water Quality Observations in the Upper Cahaba River (1970-1990)

Location code	Number of samples	Data source	Miles from H280	Miles up trib.
II, M1, B2, A3	69	S (STORET)	0	
K5	33	B (BWWB)	0	
OO, PP, QQ, RR (Trib. "100")	10	S	2.1	0.3
B6	23	B	4.0	
UT3 (Trib. "800")	16	B	9.25	0.4
UT2 (Trib. "900")	16	B	10.65	0.2
B5, CB7 (Grant's Mill Rd.)	38	B	11.15	
K4	33	B	11.35	
AC1 (Trib. "1000")	16	B	14.85	0.4
B9, K15, CB6, M2	77	B	16.10	
SC1(Stinking Creek)	16	B	17.65	1.3
A	1	S	21.35	
B (Trib. "200")	2	S	21.35	0.4
C	2	S	22.35	
E (at mouth of Big Black Cr.)	2	S	23.35	
D (Big Black Creek)	2	S	23.35	0.1
BBC1, B8	35	B	23.35	1.7
K7	31	B	23.35	4.45
K21/24	6	B	23.35	4.80
F	2	S	24.05	
G, CB5, B10	39	S/B	25.05	
K12	32	B	26.55	
H	2	S	26.95	
P	194	S	28.35	
O	2	S	28.85	
LC2 (mouth of Little Cahaba Cr.)	16	B	28.63	
Q (Little Cahaba Creek)	2	S	28.63	0.1
LC1	16	B	28.63	3.7
R	2	S	28.63	1.0
V	2	S	28.63	1.9
W	2	S	28.63	2.0
Y	2	S	28.63	2.2
I	2	S	29.73	
N	2	S	30.23	
AA	3	S	30.73	
PC2, L (mouth of Pinchgut Cr.)	19	B/S	31.33	
K (Pinchgut Creek)	5	S	31.33	0.1
J	8	S	31.33	0.4
PC1	16	B	31.33	0.5
M, K1, CB3	18	B/S	31.53	
S	10	S	32.38	
CB2, X	21	B/S	33.08	
UT1	13	B	33.88	

Appropriate water quality criteria is dependent on use classifications as stated in the *Alabama River Basin Cooperative Study Within Alabama* report (USDA and Alabama Development Office, Auburn, Alabama, April 1977, Appendix 5). The Cahaba River below the Highway 280 dam was classified for fish and wildlife uses by the Alabama Water Improvement Commission on September 17, 1973. A number of Cahaba River tributaries are also classified for swimming uses, in addition to the general fish and wildlife classification. A stretch of the river above the Highway 280 dam (to Grant's Mill Road) is also classified as a public water supply. The fish and wildlife classification includes the protection of aquatic life in the streams and the protection of human health associated with consuming fish from these waters.

The following table list the State of Alabama water quality criteria for several toxicants, from *Toxic Pollutant Criteria Applicable to State Waters* (Code of Alabama 335-6-10.07). The public water supply and swimming criteria are not shown.

	Aquatic Life Criteria		Human Life Criteria
	freshwater acute	freshwater chronic	fish consumption only
Arsenic +3	360 ug/L	190 ug/L	-
Arsenic	-	-	(1)
Cadmium	(2)	(2)	-
Chromium +3	(2)	(2)	(3)
Chromium +6	16	11	(3)
Lead	(2)	(2)	-

Mercury	2.4	0.012	(3)
Zinc	(2)	(2)	5,000 ug/L

footnotes:

(1) dependent on cancer potency and bioconcentration factors. This standard was eliminated from the State water quality criteria in April 1991.
(2) criteria dependent on water hardness.
(3) dependent on reference doses and bioconcentration factors that are developed by the EPA and used by the State of Alabama.

The Environmental Protection Agency (in *Quality Criteria for Water 1986*, EPA 440/5-86-001) recommends that the acute aquatic life criteria are for one-hour average concentrations that are not to be exceeded more than once every three years, while chronic criteria are for four-day averages that are also not to be exceeded more than once every three years.

If a large percentage of instantaneous observations (such as are contained in STORET) exceed a criterion, it is apparent, using basic statistical theory, that the observed values are not unique and that longer duration concentrations (such as the one-hour averages and the four-day averages) would also be highly likely to exceed the criterion. Therefore, the frequent exceedences reported in this report are very likely to exist at least for the durations appropriate for the various criteria.

The EPA (in *Quality Criteria for Water 1986*) uses an acceptable exceedence frequency of once per three years because they feel that three years is the average amount of time that it would take an unstressed ecosystem to recover from a pollution event in which exposure to a metal exceeds the criterion. This assumes that a population of organisms exists in adjacent unaffected areas that can recolonize the affected receiving waters. Unfortunately, many rare organisms exist in the Cahaba River that would not be able to adequately repopulate an affected area if most of the individuals are killed from a pollution incident. Therefore, even the “allowable” once-per-three-year exceedence frequency is probably too frequent to protect many of the unique and special organisms in the Cahaba River. Unfortunately, as will be shown later, many of the observed toxicant concentrations currently exceed criteria many more times than once every three years.

The EPA (also in *Water Quality Criteria*) recommends that total recoverable forms of the metals be compared to the criteria because acid soluble methods have not been approved. Most of the metal data presented in this analysis is for the filterable forms of the metals. The EPA recommended total recoverable metal forms will be greater in concentration than the filterable metal forms used in these analyses. Therefore, if the filterable metal forms exceed the criteria, it can be assumed that the total recoverable metal forms will also exceed the criteria by even larger amounts and at higher frequencies.

Water Quality Criteria for the Protection of Fish and Wildlife

The following summaries present water quality criteria to protect fish and wildlife resources. Most of this material is from the EPA’s *Water Quality Criteria* (1986) and from State of Alabama standards, with some additional notes specifically pertaining to the Cahaba River.

Dissolved Oxygen

Dissolved oxygen (DO) has received much attention as an indicator of water quality. Low levels of dissolved oxygen can produce anaerobic conditions, leading to smelly waters. Fish and other aquatic life also require suitable levels of dissolved oxygen. The oxygen requirements vary for the type of organism and its' life stage. Cold water fish are generally most sensitive, and young life forms are the most critical.

Dissolved oxygen has been a prime parameter in restricting wastewater discharges of organic material, expressed as the biochemical oxygen demand (BOD). After BOD is discharged into a receiving water, it is broken down by bacterial action. The most efficient bacteria are aerobic bacteria that consume large amounts of oxygen to stabilize organic waste discharges. In order to prevent in-stream dissolved oxygen concentrations from falling below critical levels, mathematical models are used to predict the allowable discharges of BOD for specific stream locations.

Temperature is another parameter related to dissolved oxygen. The amount of dissolved oxygen that can be contained in water (the saturation level) is dependent on the water temperature. As the water temperature increases, the saturated dissolved oxygen level decreases. The more oxygen contained in the water, the greater the waters' assimilative capacity (ability to consume organic wastes with minimal impact). Therefore, the wastewater discharges of BOD during critical summer months will have a much greater detrimental affect on stream DO than during colder months. Summer months also have lower stream flow rates, also worsening the problem by further decreasing the waters' assimilative capacity.

The EPA's national criteria for dissolved oxygen concentrations for the protection of freshwater aquatic life are presented in Table 2. These criteria were derived from the production impairment estimates which were based primarily upon growth data and information on temperature, disease, and pollutant stresses. The average dissolved oxygen concentrations selected are values 0.5 mg/L above the "slight" production impairment values and therefore represent values between no production impairment and slight production impairment. Each criterion may thus be viewed as an estimate of the threshold concentration below which detrimental effects are expected.

Table 2. Water Quality Criteria for Ambient Dissolved Oxygen Concentrations

Cold water Criteria:

	Early Life Stages[1,2]	Other Life Stages
30 day mean	NA[3]	6.5
7 day mean	9.5 (6.5)	NA
7day minimum	NA	5.0
1 day minimum[4,5]	8.0[5]	4.0

Warm water Criteria

	Early Life Stages[2]	Other Life Stages
30 day mean	NA	5.5
7 day mean	6.0	NA
7 day minimum	NA	4.0
1 day minimum	5.0	3.0

Footnotes:

1. These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. For species that have early life stages exposed directly to the water column, the figures in parentheses apply.

2. Includes all embryonic and larval stages and all juvenile forms to 30 days following hatching.

3. NA means not applicable.

4. For highly controllable discharges, further restrictions apply.

5. All minima should be considered as instantaneous concentrations to be achieved at all times.

Criteria for cold water fish are intended to apply to waters containing a population of one or more species in the family Salmonidae (Bailey, *et al*. 1970) or to waters containing other cold water or cool water fish judged to be closer to salmonids in sensitivity than to most warm water species. Although the acute lethal limit for salmonids is at or below 3 mg/L, the cold water minimum has been established at 4 mg/L because a significant proportion of the insect species common to salmonid habitats are less tolerant of acute exposures to low dissolved oxygen than are salmonids. Some cool water species may require more protection than that afforded by the other life stage criteria for warm water fish and it may be desirable to protect sensitive cool water species with the cold water criteria. Many states have more stringent dissolved oxygen standards for cooler waters, waters that contain either salmonids, nonsalmonid cool water fish, or the sensitive centrachid, the smallmouth bass The warm water criteria are necessary to protect early life stages of warm water fish as sensitive as channel catfish and to protect other life stages of fish as sensitive as largemouth bass (both occurring in the Cahaba River). Criteria for early life stages are intended to apply only where and when these life stages occur. These criteria represent dissolved oxygen concentrations which the EPA believes provide a reasonable and adequate degree of protection for freshwater aquatic life.

The criteria do not represent assured no-effect levels. However, because the criteria represent worst case conditions (i.e. for wasteload allocation and waste treatment plant design), conditions will be better than the criteria nearly all of the time at most sites. In situations where criteria conditions are just maintained for considerable periods, the criteria represent some risk of production impairment. This impairment would depend on innumerable other factors. If slight production impairment or a small but undefinable risk of moderate impairment is unacceptable, than one should use the "no production impairment" values as means and the "slight production impairment" values as minima. Table 3 presents these concentrations.

Table 3. Dissolved Oxygen Concentrations (mg/L) Versus Quantitative Level of Effect.

1. Salmonid Waters

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a. Embryo and Larval Stages

No Production Impairment = 11* (8)
Slight Production Impairment = 9* (6)
Moderate Production Impairment = 8* (5)
Severe Production Impairment = 7* (4)
Limit to Avoid Acute Mortality = 6* (3)

(* Note: These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses.)

b. Other Life Stages

No Production Impairment = 8
Light Production Impairment = 6
Moderate Production Impairment = 5
Severe Production Impairment = 4
Limit to Avoid Acute Mortality = 3

2. Nonsalmonid Waters

a. Early Life Stages

No Production Impairment = 6.5
Slight Production Impairment = 5.5
Moderate Production Impairment = 5
Severe Production Impairment = 4.5
Limit to Avoid Acute Mortality = 4

b. Other Life Stages

No Production Impairment = 6
Slight Production Impairment = 5
Moderate Production Impairment = 4
Severe Production Impairment = 3.5
Limit to Avoid Acute Mortality = 3

3. Invertebrates

No Production Impairment = 8
Some Production Impairment = 5
Acute Mortality Limit = 4

The criteria do represent dissolved oxygen concentrations believed to protect the more sensitive populations of organisms against potentially damaging production impairment. The dissolved oxygen concentrations in the criteria are intended to be protective at typically high seasonal environmental temperatures for the appropriate taxonomic and life stage classifications, temperatures which are often higher than those used in the research from which the criteria were generated, especially for other than early life stages.

Where natural conditions alone create dissolved oxygen concentrations less than 110 percent of the applicable criteria means or minima or both, the minimum acceptable concentration is 90 percent of the natural concentration. These values are similar to those presented graphically by Doudoroff and Shumway (1970) and those calculated from Water Quality Criteria 1972 (NAS/NAE 1974). Absolutely no anthropogenic dissolved oxygen depression in the potentially lethal area below the 1-day minima should be allowed unless special care is taken to ascertain the tolerance of resident species to low dissolved oxygen.

If daily cycles of dissolved oxygen are essentially sinusoidal, a reasonable daily average is calculated from the day's high and low dissolved oxygen values. A time-weighted average may be required if the dissolved oxygen cycles are decidedly non-sinusoidal. Determining the magnitude of daily dissolved oxygen cycles requires several appropriately timed measurements daily.

Once a series of daily mean dissolved oxygen concentrations are calculated, an average of these daily means can be calculated. For embryonic, larval, and early life stages, the averaging period should not exceed 7 days. This short time is needed to adequately protect these often short duration, most sensitive life stages. Other life stages can probably be adequately protected by 30-day averages. Regardless of the averaging period, the average should be considered a moving average rather than a calendar-week or calendar-month average.

A daily minimum has been included to make certain that no acute mortality of sensitive species occurs as a result of lack of oxygen. Because repeated exposure to dissolved oxygen concentrations at or near the acute lethal threshold will be stressful and because stress can indirectly produce mortality or other adverse effects (*e.g.*, through disease), the criteria are designed to prevent significant episodes of continuous or regularly recurring exposures to dissolved oxygen concentrations at or near the lethal threshold. This protection has been achieved by setting the daily minimum for early life stages at the subacute lethality threshold, by the use of a 7-day averaging period for early life stages, by stipulating a 7-day mean minimum value for other life stages, and by recommending additional limits for controllable discharges.

The previous EPA criteria for dissolved oxygen published in *Quality Criteria for Water* (USEPA 1976) was a minimum of 5 mg/L (usually applied as a 7Q10, the 7-day averaged minimum that occurs once every ten years) which is similar to the current criterion minimum except for other life stages of warm water fish which now allows a 7-day mean minimum of 4 mg/L. The new criteria are similar to those contained in the 1968 "Green Book" of the Federal Water Pollution Control Federation (FWPCA 1968).

The State of Alabama water quality criteria for dissolved oxygen is the same for fish and wildlife, and public water supply uses, the designated beneficial uses for the Upper Cahaba River:

"(i) For a diversified warm water biota, including game fish, daily dissolved oxygen concentrations shall not be less than 5 mg/L at all times; except under extreme conditions due to natural causes, it may range between 5 mg/L and 4 mg/L, provided that the water quality is favorable in all other parameters. The normal seasonal and daily fluctuations shall be maintained above these levels. In no event shall the dissolved oxygen level be less than 4 mg/L due to discharges from existing hydroelectric impoundments. All new hydroelectric generation units to existing impoundments, shall be designed so that the discharge will contain at least 5 mg/L dissolved oxygen where practicable and technologically possible. The Environmental Protection Agency, in cooperation with the State of Alabama and parties responsible for impoundments, shall develop a program to improve the design of existing facilities.

(ii) In coastal waters, surface dissolved oxygen concentrations shall not be less than 5 mg/L, except where natural phenomena cause the value to be depressed.

(iii) In estuaries and tidal tributaries, dissolved oxygen concentrations shall not be less than 5 mg/L, except in dystrophic water or where natural phenomena cause the value to be depressed.

(iv) In the application of dissolved oxygen criteria referred to above, dissolved oxygen shall be measured at a depth of 5 feet in waters 10 feet or greater in depth; and for those waters less than 10 feet in depth, dissolved oxygen criteria will be applied at mid-depth."

Bacteria

The Alabama standard for fish and wildlife are similar to the standard for a public water supply, shown in the following section, except part (i) has different limits: "Bacteria of the fecal coliform group shall not exceed a geometric mean of 1,000/100 mL on a monthly average value; nor exceed a maximum of 2,000/100 mL in any sample." Part (ii) is the same for both water beneficial uses.

Hardness

This discussion on the effects of hardness is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The water quality criteria guidance documents do not constitute a national standard, but do reflect the scientific knowledge concerning the effects of these pollutants on receiving waters.

Water hardness is caused by the divalent metallic ions (having charges of +2) dissolved in water. In fresh water, these are primarily calcium and magnesium, although other metals such as iron, strontium and manganese also contribute to the hardness content, but usually to a much lesser degree. Hardness commonly is reported as an equivalent concentration of calcium carbonate ($CaCO_3$).

Concerns about water hardness originated because hard water requires more soap to form a lather and because hard water causes scale in hot water systems. Modern use of synthetic detergents has eliminated the concern of hard water in laundries, but it is still of primary concern for many industrial water users. Many households use water softeners to reduce scale formation in hot water systems and for water taste reasons. A commonly used classification for hardness is as follows (Sawyer 1960):

Hardness concentration, mg/L as $CaCO_3$	Description
0-75	soft
75 - 150	moderately hard
150 - 300	hard
300 and up	very hard

Natural sources of hardness principally are limestones which are dissolved by percolating rainwater. Groundwaters are therefore generally harder than surface waters. Industrial sources include the inorganic chemical industry and discharges from operating and abandoned mines. Hardness in fresh water frequently is distinguished in carbonate and non-carbonate fractions. The carbonate fraction is chemically equivalent to the bicarbonates present in water. Since bicarbonates generally are measured as alkalinity, the carbonate hardness is equal to the alkalinity.

The effects of hardness on freshwater fish and other aquatic life appear to be related to the ions causing the hardness rather than by hardness as a general indicator. Both the NTAC (1968) and NAS (1974) panels have recommended against the use of the term hardness and suggested the use of the concentrations of the specific ions instead. This procedure should avoid confusion in future studies, but is not helpful in evaluating previous studies. For most existing data, it is difficult to determine whether toxicity of various metal ions is reduced because of the formation of metallic hydroxides and carbonates caused by the associated increases in alkalinity, or because of an antagonistic effect of one of the principal cations contributing to hardness, *e.g.*, calcium, or a combination of both effects. Stiff (1971) presented an example that if cupric ions were the toxic form of copper whereas copper carbonate complexes were relatively nontoxic, then the observed difference in toxicity of copper between hard and soft waters can be explained by the difference in alkalinity rather than hardness. Recent laboratory work (Engineering Foundation 1991) has also shown that alkalinity is more related to heavy metal toxicity than water hardness. As noted previously, however, carbonate hardness and alkalinity are the same.

Doudoroff and Katz (1953), in their review of the literature on toxicity, presented data showing that increasing calcium in particular reduced the toxicity of other heavy metals. Under usual conditions in fresh water and assuming that other bivalent metals behave similarly to copper, it is reasonable to assume that both effects occur simultaneously and explain the observed reduction of toxicity of metals in waters containing carbonate hardness. The amount of reduced toxicity related to hardness, as measured by a 40-hour LC5O for rainbow trout, has been estimated to be about four times for copper and zinc when the hardness was increased from 10 to 100 mg/L as $CaCO_3$ (NAS 1974). As shown in later discussions for specific heavy metals, many of the heavy metal criteria are dependent on water hardness. The allowable concentrations of cadmium, chromium, lead, and zinc to protect fish and other aquatic life, are much less in soft waters than in hard waters, for example.

Ammonia
This discussion on the effects of ammonia on aquatic life is a summary from the U.S. EPA's *Quality Criteria for Water, 1986* (EPA 1986). The criteria were published in the Federal Register (50 F.R. 30784, July 29, 1985). The ammonia criteria are only for the protection of aquatic life, as no criteria have been developed for the protection of human health (consumption of contaminated fish or drinking water). The water quality criteria is for general guidance only and do not constitute formal water quality standards. However, the criteria reflect the scientific knowledge concerning the effects of the pollutants and are recommended EPA acceptable limits for aquatic life.

All concentrations used in this EPA report are expressed as un-ionized ammonia (NH_3) because NH_3, not the ammonium ion (NH_4^+), has been demonstrated to be the principal toxic form of ammonia. The amount of the total ammonia (usually expressed as NH_3, but is really a mixture of ionized and un-ionized ammonia forms) that is un-ionized is a function of pH. At low pH values, most of the ammonia is ionized (the ammonium ion, NH_4^+), while at high pH values, most of the ammonia is un-ionized. Therefore, ammonia at high pH values creates more of a problem than similar total ammonia concentrations at low pH values. The Cahaba River watershed ammonia data reviewed is total ammonia, expressed as NH_3. The un-ionized ammonia concentrations can be calculated, if the pH values are known.

The data used in deriving the EPA criteria are predominantly from flow-through tests in which ammonia concentrations were measured. Ammonia was reported to be acutely toxic to freshwater organisms at concentrations (uncorrected for pH) ranging from 0.53 to 22.8 mg/L NH_3 for 19 invertebrate species representing 14 families and 16 genera and from 0.083 to 4.60 mg/L NH_3 for 29 fish species from 9 families and 18 genera. Among fish species, reported 96-hour LC5O values ranged from 0.083 to 1.09 mg/L for salmonids (not expected to be present in the Cahaba River) and from 0.14 to 4.60 mg/L NH_3 for nonsalmonids. Reported data from chronic tests on ammonia with two freshwater invertebrate species, both daphnids, showed effects at concentrations (uncorrected for pH) ranging from 0.304 to 1.2 mg/L NH_3, and with nine freshwater fish species, from five families and seven genera, ranging from 0.0017 to 0.612 mg/L NH_3.

Concentrations of ammonia acutely toxic to fishes may cause loss of equilibrium, hyper-excitability, increased breathing, cardiac output and oxygen uptake, and, in extreme cases, convulsions, coma, and death. At lower concentrations, ammonia has many effects on fishes, including a reduction in hatching success, reduction in growth rate and morphological development, and pathologic changes in tissues of gills, livers, and kidneys.

Several factors have been shown to modify acute NH_3 toxicity in fresh water. Some factors alter the concentration of un--ionized ammonia in the water by affecting the aqueous ammonia equilibrium, and some factors affect the toxicity of un-ionized ammonia itself, either ameliorating or exacerbating the effects of ammonia. Factors that have been shown to affect ammonia toxicity include dissolved oxygen concentration, temperature, pH, previous acclimation to ammonia, fluctuating or intermittent exposures, carbon dioxide concentration, salinity, and the presence of other toxicants.

The most well-studied of these is pH; the acute toxicity of NH_3 has been shown to increase as pH decreases. However, the percentage of the total ammonia that is un-ionized decreases with decreasing pH. Sufficient data exist from toxicity tests conducted at different pH values to formulate a relationship to describe the pH-dependent acute NH_3 toxicity. The very limited amount of data regarding effects of pH on chronic NH_3 toxicity also indicates increasing NH_3 toxicity with decreasing pH, but the data are insufficient to derive a broadly applicable toxicity/pH relationship. Data on temperature effects on acute NH_3 toxicity are limited and somewhat variable, but indications are that NH_3 toxicity to fish is greater as temperature decreases. There is no information available regarding temperature effects on chronic NH_3 toxicity.

Examination of pH and temperature-corrected acute NH_3 toxicity values among species and genera of freshwater organisms showed that invertebrates are generally more tolerant than fishes, a notable exception being the fingernail clam. There is no clear trend among groups of fish; the several most sensitive tested species and genera include representatives from diverse families (Salmonidae, Cyprinidae, Percidae, and Centrarchidae). Available chronic toxicity data for freshwater organisms also indicate invertebrates (cladocerans, one insect species) to be more tolerant than fishes, again with the exception of the fingernail clam. When corrected for the presumed effects of temperature and pH, there is also no clear trend among groups of fish for chronic toxicity values. The most sensitive species, including representatives from five families (Salmonidae, Cyprinidae, Ictaluridae, Centrarchidae, and Catostomidae), have chronic values ranging by not much more than a factor or two. Available data indicate that differences in sensitivities between warm and coldwater families of aquatic organisms are inadequate to warrant discrimination in the national ammonia criterion between bodies of water with "warm" and "coldwater" fishes; rather, effects of organism sensitivities on the criterion are most appropriately handled by site-specific criteria derivation procedures.

Data for concentrations of NH_3 toxic to freshwater phytoplankton and vascular plants, although limited, indicate that freshwater plant species are appreciably more tolerant to NH_3 than are invertebrates or fishes. The ammonia criterion appropriate for the protection of aquatic animals will therefore in all likelihood be sufficiently protective of plant life.

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The procedures described in the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if:

(1) the 1-hour* average concentration of un-ionized ammonia (in mg/L NH_3) does not exceed, more often than once every 3 years on the average, the numerical values summarized in the following table, if Salmonids and other sensitive coldwater species are absent:

One-Hour Averaged Maximum Allowable Concentrations for Total Ammonia (mg/L NH_3), For Concurrent pH and Temperature Conditions

pH	0°C	5°C	10°C	15°C	20°C	25°C	30°C
6.50	35	33	31	30	29	29	20
6.75	32	30	28	27	27	26	18.6
7.00	28	26	25	24	23	23	16.4
7.25	23	22	20	19.7	19.2	19.0	13.5
7.50	17.4	16.3	15.5	14.9	14.6	14.5	10.3
7.75	12.2	11.4	10.9	10.5	10.3	10.2	7.3
8.00	8.0	7.5	7.1	6.9	6.8	6.8	4.9
8.25	4.5	4.2	4.1	4.0	3.9	4.0	2.9
8.50	2.6	2.4	2.3	2.3	2.3	2.4	1.81
8.75	1.47	1.40	1.37	1.38	1.42	1.52	1.18
9.00	0.86	0.83	0.83	0.86	0.91	1.01	0.82

(*An averaging period of 1 hour may not be appropriate if excursions of concentrations to greater than 1.5 times the average occur during the hour; in such cases, a shorter averaging period may be needed.)

(2) the 4-day average concentration of un-ionized ammonia (in mg/L NH_3) does not exceed, more often than once every 3 years on the average, the average* numerical values summarized in the following table, if Salmonids and other sensitive coldwater species are absent:

Four-Day Averaged Maximum Allowable Concentrations for Total Ammonia (mg/L NH_3), for Concurrent pH and Temperature Conditions

pH	0°C	5°C	10°C	15°C	20°C	25°C	30°C
6.50	2.5	2.4	2.2	2.2	2.1	1.46	1.03
6.75	2.5	2.4	2.2	2.2	2.1	1.47	1.04
7.00	2.5	2.4	2.2	2.2	2.1	1.47	1.04
7.25	2.5	2.4	2.2	2.2	2.1	1.48	1.05
7.50	2.5	2.4	2.2	2.2	2.1	1.49	1.06
7.75	2.3	2.2	2.1	2.0	1.98	1.39	1.00
8.00	1.53	1.44	1.37	1.33	1.31	0.93	0.67
8.25	0.87	0.82	0.78	0.76	0.76	0.54	0.40
8.50	0.49	0.47	0.45	0.44	0.45	0.33	0.25
8.75	0.28	0.27	0.26	0.27	0.27	0.21	0.16
9.00	0.16	0.16	0.16	0.16	0.17	0.14	0.11

(*Because these criteria are nonlinear in pH and temperature, the criterion should be the average of separate evaluations of the formulas reflective of the fluctuations of flow, pH, and temperature within the averaging period; it is not appropriate in general to simply apply the formula to average pH, temperature, and flow.)

The extremes for temperature (0 and 30°C) and pH (6.5 and 9) given in the above summary tables are absolute. It is not permissible with current data to conduct any extrapolations beyond these limits. In particular, there is reason to believe that appropriate criteria at pH > 9 will be lower than the plateau between pH 8 and 9 shown above. Total ammonia concentrations equivalent to critical un-ionized ammonia concentrations are shown in these tables for receiving waters where salmonids and other sensitive coldwater species are absent, as expected for the Cahaba River. Reported EPA ammonia criteria values for salmonids and coldwater species are the same for temperatures up to 15°C. For warmer conditions, the total ammonia criteria are about 25% less.

The recommended exceedence frequency of 3 years is the EPA's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to ammonia exceeds the criterion. A stressed system, for example, one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

Nitrates

This discussion on the effects of nitrates on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). These water quality criteria guidance documents do not constitute a national standard. However, the discussion reflects the scientific knowledge concerning the effects of nitrates on the designated water uses in the Cahaba River watershed.

Two gases (molecular nitrogen and nitrous oxide) and five forms of nongaseous, combined nitrogen (amino and amide groups, ammonium, nitrite, and nitrate) are important in the nitrogen cycle. The amino and amide groups are found in soil organic matter and as constituents of plant and animal protein. The ammonium ion either is released from proteinaceous organic matter and urea, or is synthesized in industrial processes involving atmospheric nitrogen fixation. The nitrite ion is formed from the nitrate or the ammonium ions by certain microorganisms found in soil, water, sewage, and the digestive tract. The nitrate ion is formed by the complete oxidation of ammonium ions by soil or water microorganisms; nitrite is an intermediate product of this nitrification process. In oxygenated natural water systems, nitrite is rapidly oxidized to nitrate. Growing plants assimilate nitrate or ammonium ions and convert them to protein. A process known as denitrification takes place when nitrate containing soils become anaerobic and the conversion to nitrite, molecular nitrogen, or nitrous oxide occurs. Ammonium ions may also be produced in some circumstances.

Among the major point sources of nitrogen entering water bodies are municipal and industrial wastewaters, septic tanks, and feed lot discharges. Nonpoint sources of nitrogen include farm-site fertilizer and animal wastes, lawn fertilizer, sanitary landfill leachatte, atmospheric fallout, nitric oxide and nitrite discharges from automobile exhausts and other combustion processes, and losses from natural sources such as mineralization of soil organic matter (NAS 1972). Water reuse systems in some fish hatcheries employ a nitrification process for ammonia reduction; this may result in exposure of the hatchery fish to elevated levels of nitrite (Russo, *et al*. 1974).

For fingerling rainbow trout, *Salmo gairdneri*, the respective 96-hour and 7-day LC5O toxicity values were 1,360 and 1,060 mg/L nitrate nitrogen in fresh water (Westin 1974). Trama (1954) reported that the 96-hour LC5O for bluegills, *Lepomis macrochirus*, at 20°C was 2,000 mg/L nitrate nitrogen (sodium nitrate) and 420 mg/L nitrate nitrogen (potassium nitrate).

Knepp and Arkin (1973) observed that largemouth bass, *Micropterus salmoides* and channel catfish, *Ictalurus punctatus*, could be maintained at concentrations up to 400 mg/L nitrate without significant effect upon their growth and feeding activities.

Nitrite forms of nitrogen were found to be much more toxic than nitrate forms. As an example, the 96-hour and 7-day LC5O values for chinook salmon were found to be 0.9 and 0.7 mg/ L nitrite nitrogen in fresh water (Westin 1974). Smith and Williams (1974) tested the effects of nitrite nitrogen and observed that yearling rainbow trout, *Salmo gairdneri*, suffered a 55 percent mortality after 24 hours at 0.55 mg/L; fingerling rainbow trout suffered a 50 percent mortality after 24 hours of exposure at 1.6 mg/L; and chinook salmon, *Oncorhynchus tshawytscha*, suffered a 40 percent mortality within 24 hours at 0.5 mg/L. There were no mortalities among rainbow trout exposed to 0.15 mg/L nitrite nitrogen for 48 hours. These data indicate that salmonids are more sensitive to nitrite toxicity than are other fish species, e.g., minnows, *Phoxinus laevis*, that suffered a 50 percent mortality within 1.5 hours of exposure to 2,030 mg/L nitrite nitrogen, but required 14 days of exposure for mortality to occur at 10 mg/L (Klingler 1957), and carp, *Cyprinus carpio*, when raised in a water reuse system, tolerated up to 1.8 mg/L nitrite nitrogen (Saeki 1965).

The EPA concluded that (1) levels of nitrate nitrogen at or below 90 mg/L would have no adverse effects on warmwater fish (Knepp and Arkin 1973); (2) nitrite nitrogen at or below 5 mg/L should be protective of most warmwater fish (McCoy 1972); and (3) nitrite nitrogen at or below 0.06 mg/L should be protective of salmonid fishes (Russo, *et al*. 1974; Russo and Thurston 1975). These levels either are not known to occur or would be unlikely to occur in natural surface waters. Recognizing that concentrations of nitrate or nitrite that would exhibit toxic effects on warm- or coldwater fish could rarely occur in nature, restrictive criteria are not recommended.

pH
This discussion on the effects of pH is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The water quality criteria guidance documents do not constitute a national standard, but do reflect the scientific knowledge concerning the effects of these pollutants on receiving waters. State of Alabama pH standards are also discussed.

pH is a measure of the hydrogen ion activity in a water sample. It is mathematically related to hydrogen ion activity according to the expression: $pH = -\log_{10} (H^+)$, where H^+ is the hydrogen ion activity, expressed in moles/L. The pH of natural waters is a measure of the acid-base equilibrium achieved by the various dissolved compounds, salts, and gases. The principal chemical system controlling pH in natural waters is the carbonate system which is composed of atmospheric carbon dioxide (CO_2) and resulting carbonic acid (H_2CO_3), bicarbonate ions (HCO_3^-) and carbonate ions (CO_3^{2-}) The interactions and kinetics of this system have been described by Stumm and Morgan (1970).

pH is an important factor in the chemical and biological reactions in natural waters. The degree of dissociation of weak acids or bases is affected by changes in pH. This effect is important because the toxicity of many compounds is affected by the degree of dissociation. One such example is for hydrogen cyanide. Cyanide toxicity to fish increases as the pH is lowered because the chemical equilibrium is shifted towards an increased concentration of a more toxic form of cyanide. Similar results have also been shown for hydrogen sulfide (H_2S) (Jones 1964). Conversely, rapid increases in pH can cause increased NH_3 concentrations that are also toxic. Ammonia has been shown to be 10 times as toxic at pH 8.0 as at pH 7.0 (EIFAC 1969).

The solubility of metal compounds contained in bottom sediments, or as suspended material, also is affected by pH. For example, laboratory equilibrium studies under anaerobic conditions indicated that pH was an important parameter involved in releasing manganese from bottom sediments (Delfino and Lee 1971).

Knowledge of pH in the raw water used for public water supplies is important because without adjustment to a suitable level, such waters may be corrosive and adversely affect treatment processes, especially coagulation and chlorination.

Coagulation, used for removal of colloidal color and turbidity through the use of aluminum or iron salts, generally has an optimum pH range of 5.0 to 6.5 (Sawyer 1960). The effect of pH on chlorine in water principally concerns the equilibrium between hypochlorous acid (HOCl) and the hypochlorite ion (OCl^-) according to the reaction:

$$HOCI = H^+ + OCI^-$$

High hydrogen ion concentrations (low pH) would therefore cause much more HOCl to be present, than at high pH values. Butterfield (1984) has shown that chlorine disinfection is more effective at values less than pH 7 (favoring HOCl, the more effective disinfectant). Water is therefore adjusted to a pH of between 6.5 and 7 before most water treatment processes.

Corrosion of plant equipment and piping in the distribution system can lead to expensive replacement as well as the introduction of metal ions such as copper, lead, zinc, and cadmium. Langelier (1936) developed a method to calculate and control water corrosive activity that employs calcium carbonate saturation theory and predicts whether the water would tend to dissolve metal piping, or deposit a protective layer of calcium carbonate on the metal. Generally, this level is above pH 7 and frequently approaches pH 8.3, the point of maximum bicarbonate/carbonate buffering.

Since pH is relatively easily adjusted prior to, and during, water treatment, a rather wide range is acceptable for waters serving as a source of public water supply. A range of pH from 5.0 to 9.0 would provide a water treatable by typical (coagulation, sedimentation, filtration , and chlorination) treatment plant processes. As the range is extended, the cost of pH adjusting chemicals increases.

A review of the effects of pH on fresh water fish has been published by the European Inland Fisheries Advisory Commission (1969). The commission concluded:

There is no definite pH range within which a fishery is unharmed and outside which it is damaged, but rather, there is a gradual deterioration as the pH values are further removed from the normal range. The pH range which is not directly lethal to fish is 5 to 9; however, the toxicity of several common pollutants is markedly affected by pH changes within this range, and increasing acidity or alkalinity may make these poisons more toxic. Also, an acid discharge may liberate sufficient CO_2 from bicarbonate in the water either to be directly toxic, or to cause the pH range of 5 to 6 to become lethal.

Mount (1973) performed bioassays on the fathead minnow, *Pimephales promelas*, for a 13-month, one generation time period to determine chronic pH effects. Tests were run at pH levels of 4.5, 5.2, 5.9, 6.6, and a control of 7.5. At the two lowest pH values (4.5 and 5.2) behavior was abnormal and the fish were deformed. At pH values less than 6.6, egg production and hatchability were reduced when compared with the control. It was concluded that a pH of 6.6 was marginal for vital life functions. Bell (1971) performed bioassays with nymphs of caddisflies (two species) stoneflies (four species), dragonflies (two species), and mayflies (one species). All are important fish food organisms. The 30-day TL5O pH values ranged from 2.5 to 5.4, with the caddisflies being the most tolerant and the mayflies being the least tolerant. The pH values at which 50 percent of the organisms emerged ranged from 4.0 to 6.6 with increasing percentage emergence occurring with the increasing pH values.

Based on present evidence, a pH range of 6.5 to 9.0 appears to provide adequate protection for the life of freshwater fish and bottom dwelling invertebrates. Outside of this range, fish suffer adverse physiological effects increasing in severity as the degree of deviation increases until lethal levels are reached:

pH Range	Effect on Fish
5.0 - 6.0	Unlikely to be harmful to any species unless either the concentration of free CO_2 is greater than 20 ppm, or the water contains iron salts which are precipitated as ferric hydroxide, the toxicity of which is not known.
6.0 - 6.5	Unlikely to be harmful to fish unless free CO_2 is present in excess of 100 ppm.
6.5 - 9.0	Harmless to fish, although the toxicity of other poisons may be affected by changes within this range.

source: EIFAC 1969

The EPA recommended water quality criteria for pH therefore restricts pH values to be in the range of 5 to 9 for domestic water supplies (welfare), and within the range of 6.5 to 9.0 for freshwater aquatic life protection. The State of Alabama's fresh water pH standards for public water supplies and aquatic life are: "Sewage, industrial wastes or other wastes shall not cause the pH to deviate more than one unit from the normal or natural pH, nor be less than 6.0, nor greater than 8.5."

Phosphate

This discussion on the effects of phosphate on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The phosphate observations for the Cahaba River study area are for total forms of the nutrient. These water quality criteria guidance documents do not constitute a national standard. However, the discussion reflects the scientific knowledge concerning the effects of phosphates on the designated water uses in the Cahaba River watershed.

Phosphorus in the elemental form is very toxic (having an EPA marine life criteria of 0.10 μg/L) and is subject to bioaccumulation in much the same way as mercury. Phosphate forms of phosphorus are a major nutrient required for plant

nutrition. In excessive concentrations, phosphates can stimulate plant growth. Excessive growths of aquatic plants (eutrophication) often interfere with water uses and are nuisances to man. Generally, phosphates are not the only cause of eutrophication, but there is substantiating evidence that frequently it is the key element of all of the elements required by freshwater plants (generally, it is present in the least amount relative to need). Therefore, an increase in phosphorus allows use of other already present nutrients for plant growth. In addition, of all of the elements required for plant growth in the water environment, phosphorus is the most easily controlled by man.
However, in most parts of the Cahaba River basin, nitrogen compounds are likely the most critical nutrients because of the relatively large amounts of treated sewage, which is especially high in phosphates, in relation to other pollution sources.

Phosphates enter waterways from several different sources. The human body excretes about one pound per year of phosphorus compounds. The use of phosphate detergents increases the per capita contribution to about 3.5 pounds per year of phosphorus compounds. Some industries, such as potato processing, have wastewaters high in phosphates. Many non-point sources (crop, forest, idle, and urban lands) contribute varying amounts of phosphorus compounds to watercourses. This drainage may be surface runoff of rainfall, effluent from agricultural tile lines, or return flow from irrigation. Cattle feedlots, birds, tree leaves, and fallout from the atmosphere all are contributing sources.

Evidence indicates that: (1) high phosphorus compound concentrations are associated with accelerated eutrophication of waters, when other growth-promoting factors are present; (2) aquatic plant problems develop in reservoirs and other standing waters at phosphorus values lower than those critical in flowing streams; (3) reservoirs and lakes collect phosphates from influent streams and store a portion of them within consolidated sediments, thus serving as a phosphate sink; and (4) phosphorus concentrations critical to noxious plant growth vary and nuisance growths may result from a particular concentration of phosphate in one geographical area but not in another. The amount or percentage of inflowing nutrients that may be retained by a lake or reservoir is variable and will depend upon: (1) the nutrient loading to the lake or reservoir; (2) the volume of the euphotic zone; (3) the extent of biological activities; (4) the detention time within a lake basin or the time available for biological activities; and (5) the discharge from the lake.

Once nutrients are discharged into an aquatic ecosystem, their removal is tedious and expensive. Phosphates are used by algae and higher aquatic plants and may be stored in excess of use within the plant cells. With decomposition of the plant cell, some phosphorus may be released immediately through bacterial action for recycling within the biotic community, while the remainder may be deposited with sediments. Much of the material that combines with the consolidated sediments within the lake bottom is bound permanently and will not be recycled into the system.

Although a total phosphorus criterion to control nuisance aquatic growths is not presented, the EPA believes that the following rationale to support such a criterion, which currently is evolving, should be considered.

Total phosphate concentrations in excess of 100 μg/L (expressed as total phosphorus) may interfere with coagulation in water treatment plants. When such concentrations exceed 25 μg/L at the time of the spring turnover on a volume-weighted basis in lakes or reservoirs, they may occasionally stimulate excessive or nuisance growths of algae and other aquatic plants. Algal growths cause undesirable tastes and odors to water, interfere with water treatment, become aesthetically unpleasant, and alter the chemistry of the water supply. They contribute to eutrophication.

To prevent the development of biological nuisances and to control accelerated or cultural eutrophication, total phosphates as phosphorus (P) should not exceed 50 μg/L in any stream at the point where it enters any lake or reservoir, nor 25 μg/L within the lake or reservoir. A desired goal for the prevention of plant nuisances in streams or other flowing waters not discharging directly to lakes or impoundments is 100 μg/L total P (Mackenthun 1973). Most relatively uncontaminated lake districts are known to have surface waters that contain from 10 to 30 μg/L total phosphorus as P (Hutchinson, 1957).

The majority of the Nation's eutrophication problems are associated with lakes or reservoirs and currently there are more data to support the establishment of a limiting phosphorus level in those waters than in streams or rivers that do not directly impact such water. There are natural conditions, also, that would dictate the consideration of either a more or less stringent phosphorus level. Eutrophication problems may occur in waters where the phosphorus concentration is less than that indicated above and, obviously, such waters would need more stringent nutrient limits. Likewise, there are those waters within the Nation where phosphorus is not now a limiting nutrient and where the need for phosphorus limits is substantially diminished.

It is evident that a portion of that phosphorus that enters a stream or other flowing waterway eventually will reach a receiving lake or estuary either as a component of the fluid mass, as bed load sediments that are carried downstream, or as floating organic materials that may drift just above the stream's bed or float on its water's surface. Superimposed on the loading from the inflowing waterway, a lake or estuary may receive additional phosphorus as fallout from the atmosphere or as a direct introduction from shoreline areas.

Another method to control the inflow of nutrients, particularly phosphates, into a lake is that of prescribing an annual loading to the receiving water. Vollenweider (1973) suggests total phosphorus (P) loadings, in grams per square meter of surface area per year, that will be a critical level for eutrophic conditions within the receiving waterway for a particular water volume. The mean depth of the lake in meters is divided by the hydraulic detention time in years. Vollenweider's data suggest a range of loading values that should result in oligotrophic lake water quality:

Mean Depth/Hydraulic Detention Time (meters/year)	Oligotrophic or Permissible Loading (grams/meter/year)	Eutrophic or Critical Loading (grams/meter/year)
0.5	0.07	0.14
1.0	0.10	0.20
2.5	0.16	0.32
5.0	0.22	0.45
7.5	0.27	0.55
10.0	0.32	0.63
25.0	0.50	1.00
50.0	0.71	1.41
75.0	0.87	1.73
100.0	1.00	2.00

There may be waterways where higher concentrations, or loadings, of total phosphorus do not produce eutrophication, as well as those waterways where lower concentrations or loadings of total phosphorus may be associated with populations of nuisance organisms. Waters now containing less than the specified amounts of phosphorus should not be degraded by the introduction of additional phosphates

It should be recognized that a number of specific exceptions can occur to reduce the threat of phosphorus as a contributor to lake eutrophication:

1. Naturally occurring phenomena may limit the development of plant nuisances.
2. Technological or cost effective limitations may help control introduced pollutants.
3. Waters may be highly laden with natural silts or colors which reduce the penetration of sunlight needed for plant photosynthesis.
4. Some waters physical features of steep banks, great depth, and substantial flows contribute to a history of no plant problems.
5. Waters may be managed primarily for waterfowl or other wildlife.
6. In some waters, nutrients other than phosphorus (such as nitrogen) is limiting to plant growth; the level and nature of such limiting nutrient would not be expected to increase to an extent that would influence eutrophication.
7. In some waters, phosphorus control cannot be sufficiently effective under present technology to make phosphorus the limiting nutrient.

Dissolved Solids, Conductivity, and Chlorides
This discussion on the effects of total dissolved solids, chlorides, and conductivity on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The water quality criteria guidance documents do not constitute a national standard, but do reflect the scientific knowledge concerning the effects of these pollutants on receiving waters.

Total dissolved solids, chlorides, and conductivity observations are typically used to indicate the magnitude of dissolved minerals in the water. The term total dissolved solids (or dissolved solids) is generally associated with freshwater and refers to the inorganic salts, small amounts of organic matter, and dissolved materials in the water (Sawyer 1960). Salinity is an oceanographic term, and although not precisely equivalent to the total dissolved salt content, it is related (Capurro 1970). Chlorides (not chlorine) are directly related to salinity because of the constant relationship between the major salts in sea water. Conductivity is a measure of the electrical conductivity of water and is also generally related to total dissolved solids, chlorides, or salinity. The principal inorganic anions (negatively charged ions) dissolved in fresh water include the carbonates, chlorides, sulfates, and nitrates (principally in groundwaters); the principal cations (positively charged ions) are sodium, potassium, calcium, and magnesium.

All species of fish and other aquatic life must tolerate a range of dissolved solids concentrations in order to survive under natural conditions. Studies in Saskatchewan found that several common freshwater species survived 10,000 mg/L dissolved solids, that whitefish and pikeperch survived 15,000 mg/L, but only the stickleback survived 20,000 mg/L dissolved solids. It was concluded that lakes with dissolved solids in excess of 15,000 mg/L were unsuitable for most freshwater fishes (Rawson and Moore 1944). The 1968 NTAC Report also recommended maintaining osmotic pressure levels of less than that caused by a 15,000 mg/L solution of sodium chloride.

Indirect effects of excess dissolved solids are primarily the elimination of desirable food plants and other habitat-forming plants. Rapid salinity changes cause plasmolysis of tender leaves and stems because of changes in osmotic pressure. The 1968 NTAC Report recommended the following limits in salinity variation from natural to protect wildlife habitats:

Natural Salinity (parts per thousand)	Variation Permitted (parts per thousand)
0 to 3.5 (freshwater)	1
3.5 to 13.5 (brackish water)	2
13.5 to 35 (seawater)	4

The State of Alabama has used a chloride criteria of 230 mg/L to protect aquatic life in the Cahaba River.

Temperature

This discussion on the effects of temperature is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The water quality criteria guidance documents do not constitute a national standard, but do reflect the scientific knowledge concerning the effects of these pollutants on receiving waters. State of Alabama temperature standards are also discussed.

Water temperature affects many beneficial uses, including industrial and domestic water supplies and recreation. The effects of temperature on aquatic life are of the most concern, however, and the water quality criteria were developed to protect the most sensitive aquatic organisms from stress associated with elevated temperatures. Since essentially all of the aquatic organisms are cold blooded, the temperature of the water regulates their metabolism and their ability to survive and reproduce. Temperature, therefore, is an important physical parameter which to some extent regulates many of the beneficial uses of water. The Federal Water Pollution Control Administration in 1967 called temperature "a catalyst, a depressant, an activator, a restrictor, a stimulator, a controller, a killer, one of the most important and most influential water quality characteristics to life in water."

The suitability of water for total body immersion is greatly affected by temperature. In temperate climates, dangers from exposure to low temperatures is more prevalent than exposure to elevated water temperatures. Depending on the amount of activity by the swimmer, comfortable temperatures range from 20^{o} C to 30^{o} C. Short durations of lower and higher temperatures can be tolerated by most individuals. For example, for a 30-minute period, temperatures of 10^{o} C or 35^{o} C can be tolerated without harm by most individuals (NAS 1974).

Temperature also affects the self-purification phenomenon in water bodies and therefore the aesthetic and sanitary qualities that exist. Increased temperatures accelerate the biodegradation of organic material both in the overlying water and in bottom deposits which makes increased demands on the dissolved oxygen resources of a given system. The typical situation is exacerbated by the fact that oxygen becomes less soluble as water temperature increases. Thus, greater demands are exerted on an increasingly scarce resource which may lead to total oxygen depletion and obnoxious septic conditions.

Temperature changes in water bodies can alter the existing aquatic community. The dominance of various phytoplankton groups in specific temperature ranges has been shown. For example, from 20^{o} C to 25^{o} C, diatoms predominated; green algae predominated from 30^{o} C; to 35^{o} C and blue-greens predominated above 35^{o} C (Cairns 1956). Likewise, changes from a coldwater fishery to a warm-water fishery can occur because temperature may be directly lethal to adults or fry, or cause a reduction of activity, or limit their reproduction (Brett 1969).

Upper and lower limits for temperature have been established for many aquatic organisms. Considerably more data exist for upper, as opposed to lower limits. Tabulations of lethal temperatures for fish and other organisms are available (Jones 1964; FWPCA 1967; NAS 1974). Factors such as diet, activity, age, general health, osmotic stress, and even weather contribute to the lethality of temperature. The aquatic species and exposure time are considered the critical factors (Parker and Krenkel 1969).

The effects of sublethal temperatures on metabolism, respiration, behavior, distribution and migration, feeding rate, growth, and reproduction have been summarized by De Sylva (1969). Another study has illustrated that inside the tolerance zone, there is a more restrictive temperature range in which normal activity and growth occur and yet an even more restrictive zone in which normal reproduction will be occur (Brett 1960).

De Sylva (1969) has summarized available data on the combined effects of increased temperature and toxic materials on fish. These data indicate that toxicity generally increases with increased temperature and that organisms subjected to stress from toxic materials are less tolerant of temperature extremes.

The tolerance of organisms to extremes of temperature is a function of their genetic ability to adapt to thermal changes within their characteristic temperature range, the acclimation temperature prior to exposure, and the time of exposure to the elevated temperature (Coutant 1972). True acclimation to changing temperatures requires several days (Brett 1941). Organisms that are acclimated to relatively warm water, when subjected to reduced temperatures that under other conditions of acclimation would not be detrimental, may suffer significant mortality caused by thermal shock (Coutant 1972).

Through the natural changes in climatic conditions, the temperatures of water bodies fluctuate daily, as well as seasonally. These changes do not eliminate indigenous aquatic populations, but affect the existing community structure and the geographic distribution of species. Such temperature changes are necessary to induce the reproductive cycles of aquatic organisms and to regulate other life factors (Mount 1969).

In open waters elevated temperatures may affect periphyton, benthic invertebrates, and fish, in addition to causing shifts in algal dominance. Trembley (1960) studies of the Delaware River downstream from a power plant concluded that the periphyton population was considerably altered by the discharge.

The number and distribution of bottom organisms decrease as water temperatures increase. The upper tolerance limit for a balanced benthic population structure is approximately 32^{o} C. A large number of these invertebrate species are able to tolerate higher temperatures than those required for reproduction (FWPCA 1967).

In order to define criteria for fresh waters, Coutant (1972) cited the following as definable requirements:

1. Maximum sustained temperatures that are consistent with maintaining desirable levels of productivity.
2. Maximum levels of metabolic acclimation to warm temperatures that will permit return to ambient winter temperatures should artificial sources of heat cease.
3. Time-dependent temperature limitations for survival of brief exposures to temperature extremes, both upper and lower.
4. Restricted temperature ranges for various states of reproduction, including (for fish) gametogenesis, spawning migration, release of gametes, development of the embryo, commencement of independent feeding (and other activities) by juveniles, and temperatures required for metamorphosis, emergence, or other activities of lower forms.
5. Thermal limits for diverse species compositions of aquatic communities, particularly where reduction in diversity creates nuisance growths of certain organisms, or where important food sources (food chains) are altered,
6. Thermal requirements of downstream aquatic life (in rivers) where upstream flow reductions of a coldwater resource will adversely affect downstream temperature requirements.

To provide a safety factor, so that none, or only a few, organisms will perish, it has been found experimentally that a criterion of 2^{o} C below maximum temperature is usually sufficient (Black 1953). To provide safety for all the organisms, the temperature causing a median mortality for 50 percent of the population should be calculated and reduced by 2^{o} C in the case of an elevated temperature.

Maximum temperatures for an extensive exposure (e.g., more than 1 week) must be divided into those for warmer periods and winter. Other than for reproduction, the most temperature sensitive life function appears to be growth (Coutant 1972). Coutant (1972) has suggested that a satisfactory estimate of a limiting maximum weekly mean temperature may be an average of the optimum temperature for growth and the temperature for zero net growth.

Because of the difficulty in determining the temperature of zero net growth, essentially the same temperature can be derived by adding to the optimum temperature (for growth or other physiological functions) a factor calculated as onethird of the difference between the ultimate upper lethal temperature and the optimum temperature (NAS 1974).

Since temperature tolerance varies with various states of development of a particular species, the criterion for a particular location should be calculated for the most important life form likely to be present during a particular month. One caveat in using the maximum weekly mean temperature is that the limit for short-term exposure must not be exceeded. Example calculations for predicting the summer maximum temperatures for short-term survival and for extensive exposure for various fish species are presented in Table 4. These values use data from EPA's Environmental Research Laboratory (ERL) in Duluth.

Table 4. Maximum Weekly Average Temperatures for Growth, and Short-Term Maxima for Survival for Juveniles and Adults During the Summer (Centigrade and Fahrenheit)

Species[a]	Growth[b]	Maxima[c]
Bluegill	32 (90)	35 (95)
Channel catfish	32 (90)	35 (95)
Largemouth bass	32 (90)	34 (93)

a - These species were found in the upper Cahaba River (Pierson, *et al.* 1989).
b - Calculated using optimum temperature for growth: maximum weekly average temperature for growth = optimum temperature + 1/3 (ultimate lethal temperature - optimum temperature).
c - Based on acclimation temperature, at the maximum weekly average temperature, needed for summer growth, minus 2^{o} C.

The winter maximum temperature must not exceed the ambient water temperature by more than the amount of change a specimen acclimated to a discharge temperature can tolerate. Such a change could occur by a cessation of the source of heat or by the specimen being driven from an area by high flows, pollutants, or other factors. However, there are inadequate data to estimate a safety factor for the "no stress" level from cold shocks (NAS 1974).

Coutant (1972) has reviewed the effects of temperature on aquatic life reproduction and development. Reproductive events are noted as perhaps the most thermally restricted of all life phases assuming other factors are at or near optimum levels. Natural short-term temperature fluctuations appear to cause reduced reproduction of fish and invertebrates.

There are inadequate data available quantifying the most temperature sensitive life stages among various aquatic species. Uniform elevation of temperature a few degrees, but still within the spawning range, may lead to advanced spawning for spring spawning species and delays for fall spawners. Such changes may not be detrimental, unless asynchrony occurs between newly hatched juveniles and their normal food source. Such asynchrony may be most pronounced among anadromous species, or other migrants, who pass from the warmed area to a normally chilled, unproductive area. Reported temperature data on maximum temperatures for spawning and embryo survival have been summarized in Table 5 (from ERL-Duluth 1976).

Table 5. Maximum Weekly Average Temperatures for Spawning and Short-Term Maxima for Embryo Survival During Spawning Season (Centigrade and Fahrenheit)

Species[a]	Spawning[b]	Survival[c]
Bluegill	25 (77)	34 (93)
Channel catfish	27 (81)	29 (84)
Largemouth bass	21 (70)	27 (81)
Threadfin shad	18 (64)	34 (93)

a - These species were found in the upper Cahaba River (Pierson, *et al.* 1989).
b - The optimum, or mean of the range, of spawning temperatures reported for the species (ERL-Duluth 1976).
c - The upper temperature for successful incubation and hatching reported for the species (ERL-Duluth 1976).

The recommended EPA criteria is in two main parts. The second part is also broken down into four subparts. This detail is needed to account for the differences in temperature tolerance for various aquatic organisms. The EPA criteria are as follows:

For any time of year, there are two upper limiting temperatures for a location (based on the important sensitive species found there at that time):

1. One limit consists of a maximum temperature for short exposures that is time dependent and is given by the species specific equation (example calculated values are shown on Table 5 under the “maxima” column):

Temperature =(1/b)[log (time) -a] -2^{o} C

where: Temperature is ^{o} C,
exposure time is in minutes,
a= intercept on the “y” or logarithmic axis of the line fitted to experimental data and which is available for some species from Appendix Il-C, National Academy of Sciences 1974 document.
b= slope of the line fitted to experimental data and available for some species from Appendix Il-C, of the National Academy of Sciences 1974 document.

2. The second value is a limit on the weekly average temperature that:

a. In the cooler months (mid-October to mid-April in the north and December to February in the south) will protect against mortality of important species if the elevated plume temperature is suddenly dropped to the ambient temperature, with the limit being the acclimation temperature minus 2^{o} C when the lower lethal threshold temperature equals the ambient water temperature (in some regions this limitation may also be applicable in summer). or

b. In the warmer months (April through October in the north and March through November in the south) is determined by adding to the physiological optimum temperature (usually for growth) a factor calculated as one-third of the difference between the ultimate upper lethal temperature and the optimum temperature for the most sensitive important species (and appropriate life state) that normally is found at that location and time. (Some of these values are given in Table 5 under the “growth” column). or

c. During reproductive seasons (generally April through June and September through October in the north and March through May and October through November in the south) the limit is that temperature that meets site - specific requirements for successful migration, spawning, egg incubation, fry rearing, and other reproductive functions of important species. These local requirements should supersede all other requirements when they are applicable. or

d. There is a site-specific limit that is found necessary to preserve normal species diversity or prevent appearance of nuisance organisms.

The most critical temperatures for the limited data available for upper Cahaba River fish are 34^{o}C (Largemouth bass - maxima, all times), 32^{o}C (Bluegill, Channel catfish, and largemouth bass - growth, March through November), 27^{o}C (Largemouth bass - embryo survival, October and November), and 18^{o}C (Threadfin shad - spawning, October and November).

The State of Alabama has the same temperature water quality standards for both public water supplies and for the protection of fish and other aquatic organisms. These standards (potentially affecting the Cahaba River) are:

(i) The maximum temperature in streams, lakes and reservoirs, other than those in river basins listed in subparagraph (ii) hereof, shall not exceed 90^{o}F.
(ii) The maximum temperature in streams, lakes and reservoirs in the Tennessee and Cahaba River Basins, and for that portion of the Tallapoosa River Basin from the tailrace of Thurlow Dam at Tallassee downstream to the junction of the Coosa and Tallapoosa Rivers which has been designated by the Alabama Department of Conservation and Natural Resources as supporting smallmouth bass, sauger, or walleye, shall not exceed 86^{o}F.
(iii) The maximum in-stream temperature rise above ambient water temperature due to the addition of artificial heat by a discharger shall not exceed 5^{o}F in streams, lakes and reservoirs in non-coastal and non-estuarine areas.
(v) In lakes or reservoirs there shall be no withdrawal from, nor discharge of heated waters to, the hypolimnion unless it can be shown that such discharge will be beneficial to water quality.
(vi) In all waters the normal daily and seasonal temperature variations that were present before the addition

of artificial heat shall be maintained, and there shall be no thermal block to the migration of aquatic organisms.

Suspended Solids and Turbidity
This discussion on the effects of suspended solids and turbidity on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). These water quality criteria guidance documents do not constitute a national standard. However, the discussion reflects the scientific knowledge concerning the effects of suspended solids and turbidity on the designated water uses in the Cahaba River watershed. Alabama State standards for turbidity are also discussed.

Suspended solids (sometimes referred to as nonfilterable residue) and turbidity are related to the solids content of the water that is not dissolved. Turbidity refers to the blockage of light penetration and is measured by examining the backscatter from an intense light beam, while suspended solids is measured by weighing the amount of dried sediment that is trapped on a 0.45 micron filter, after filtering a known sample volume. The suspended solids test therefore measures a broad variety of solids that are contained in the wastewater, including floatable material and settleable matter, in addition to the suspended solids. An Imhoff cone can be used to qualitatively estimate the settleable solids content of a wastewater. Subjecting the filter to a high temperature will burn off the more combustible solids. The remaining solids is usually referred to as the nonvolatile solids. The amount burned is assumed to be related to the organic fraction of the wastewater.

Turbidity (and color) can be mostly caused by very small particles (less than 1 μm), while the suspended solids content is usually associated with more moderate sized particles (10 to 100 μm). Suspended solids can cause water quality problems directly, as discussed in the following paragraphs from *Water Quality Criteria* (1986). They may also have other pollutants (such as organics and toxicants) associated with them that would cause additional problems. The control of suspended solids is required in most discharge permits because of potential sedimentation problems downstream of the discharge and the desire to control associated other pollutants.

Turbid water interferes with recreational use and aesthetic enjoyment of water. Turbid waters can be dangerous for swimming, especially if diving facilities are provided, because of the possibility of unseen submerged hazards and the difficulty in locating swimmers in danger of drowning (NAS 1974). The less turbid the water, the more desirable it becomes for swimming and other water contact sports. Other recreational pursuits, such as boating and fishing, will be adequately protected by suspended solids criteria developed for protection of fish and other aquatic life.

Fish and other aquatic life requirements concerning suspended solids can be divided into those whose effect occurs in the water column and those whose effect occurs following sedimentation to the bottom of the water body. Noted effects are similar for both fresh and marine waters.

The effects of suspended solids on fish have been reviewed by the European Inland Fisheries Advisory Commission (EIFAC 1965). This review in 1965 identified four effects on the fish and fish food populations, namely:

(1) By acting directly on the fish swimming in water in which solids are suspended, and either killing them or reducing their growth rate, resistance to disease, etc.;
(2) by preventing the successful development of fish eggs and larvae;
(3) by modifying natural movements and migrations of fish; and
(4) by reducing the abundance of food available to the fish.

Settleable materials which blanket the bottom of water bodies damage the invertebrate populations, block gravel spawning beds, and if organic, remove dissolved oxygen from overlying waters (EIFAC 1965; Edberg and Hofsten 1973). In a study downstream from the discharge of a rock quarry where inert suspended solids were increased to 80 mg/L, the density of macroinvertebrates decreased by 60 percent while in areas of sediment accumulation, benthic invertebrate populations also decreased by 60 percent regardless of the suspended solid concentrations (Gammon 1970). Similar effects have been reported downstream from an area which was intensively logged. Major increases in stream suspended solids (25 mg/L upstream versus 390 mg/L downstream) caused smothering of bottom invertebrates, reducing organism density to only 7.3 per square foot versus 25.5 per square foot upstream (Tebo 1955).

Deposition of organic materials to the bottom sediments can cause imbalances in stream biota by increasing bottom animal density (principally worms), and diversity is reduced as pollution-sensitive forms disappear (Mackenthun 1973). Algae, likewise, flourish in such nutrient-rich areas, although forms may become less desirable (Tarzwell and Gaufin 1953).

Plankton and inorganic suspended materials reduce light penetration into the water body, reducing the depth of the photic zone. This reduces primary production and decreases fish food. The NAS committee in 1974 recommended that the depth of light penetration not be reduced by more than 10 percent (NAS 1974). Additionally, the near surface waters are heated because of the greater heat absorbency of the particulate material which tends to stabilize the water column and prevents vertical mixing (NAS 1974). Such mixing reductions decrease the dispersion of dissolved oxygen and nutrients to lower portions of the water body. Increased temperatures also reduce the capacity of the stream to contain dissolved oxygen.

Suspended inorganic material in water also sorbs organic materials, such as pesticides. Following this sorption process, subsequent sedimentation may remove these materials from the water column into the sediments (NAS 1974). However, the sedimentation of these polluted sediments can cause dramatic changes in the benthic microorganism populations, which in turn affect other aquatic life forms. More recent research associated with the effects of polluted sediments in urban streams is summarized by Pitt (1991).

The EPA water quality criterion for freshwater fish and other aquatic life are essentially that proposed by the National Academy of Sciences and the Great Lakes Water Quality Board: "Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life." The state of Alabama water quality criterion for turbidity is the same for all designated uses: "There shall be no turbidity of other than natural origin that will cause substantial visible contrast with the natural appearance of waters or interfere with any beneficial uses which they serve. Furthermore, in no case shall turbidity exceed 50 Nephelometric units (NTU) above background. Background will be interpreted as the natural condition of the receiving waters, without the influence of man-made or man-induced causes. Turbidity levels caused by natural runoff will be included in establishing background levels." In addition, the state of Alabama has minimum conditions applicable to all state waters that includes: "State waters shall be free from substances attributable to sewage, industrial wastes or other wastes that will settle to form bottom deposits which are unsightly, putrescent or interfere directly or indirectly with any classified water use."

Heavy Metals

The State of Alabama has established water quality criteria for various heavy metals for fish and wildlife protection, the common designated uses of the Cahaba River. Many of the criteria shown above are defined in terms of water hardness, as elevated water hardness levels have been demonstrated in many laboratory experiments to lessen the toxic effects of these metals. Water hardness values in the Cahaba River were therefore examined (as presented in the STORET records) for the Cahaba River at Centreville. The following list shows the percentage of the 71 observations that were less than the hardness values indicated:

percentile	water hardness (mg/L as $CaCO_3$)
0% (minimum)	25
10	42
20	54
30	63
40	74
50 (median)	84
60	90
70	98
80	110
90	120
100 (maximum)	140

These percentile values were then used in the equations presented in the Alabama *Toxic Pollutant Criteria Applicable to State Waters* (Code of Alabama 335-6-10.07). The following tables summarize the applicable criteria, associated with each percentile value of hardness:

Alabama Freshwater Aquatic Life Criteria (μg/L)

percent	hardness	Cadmium		Chromium(+3)	
	mg/L	acute	chronic	acute	chronic
0%	25	0.82	0.38	560	67
10	42	1.5	0.57	850	100
20	54	2.0	0.70	1050	125
30	63	2.3	0.79	1190	140
40	74	2.8	0.90	1360	160
50	84	3.2	0.99	1500	180

60	90	3.5	1.0	1590	190
70	98	3.8	1.1	1710	200
80	110	4.4	1.2	1880	220
90	120	4.8	1.3	2020	240
100	140	5.7	1.5	2290	270

Alabama Freshwater Aquatic Life Criteria (μg/L) (Cont.)

percent	hardness	Lead		Zinc	
	mg/L	acute	chronic	acute	chronic
0%	25	14	0.54	36	33
10	42	27	1.1	56	51
20	54	37	1.5	69	63
30	63	45	1.8	79	72
40	74	56	2.2	91	82
50	84	65	2.5	100	91
60	90	71	2.8	110	97
70	98	80	3.1	115	100
80	110	92	3.6	130	115
90	120	100	4.0	140	120
100	140	125	4.9	160	140

Hexavalent chromium (Cr^{+6}) and mercury aquatic life problems are not effected by hardness and the State of Alabama has established the following criteria to protect aquatic life from exposure to these two metals:

Mercury acute criterion: 2.4 μg/L
Mercury chronic criterion: 0.012 μg/L
Chromium +6 acute criterion: 16 μg/L
Chromium +6 chronic criterion: 11 μg/L

As noted above, the EPA suggests that these aquatic life criteria should not be exceeded more than once every three years. The acute criteria is for a one-hour average, while the chronic criteria is for a four-day average.

Water Quality Criteria for the Protection of Human Health

The following discussion is mostly from the EPA's *Water Quality Criteria* (1986), and applicable state of Alabama regulations. It summarizes applicable water quality criteria for the protection of human health through both drinking water and fish consumption pathways. Water contact recreation is also considered for bacteria.

Bacteria

A recreational water quality criterion can be defined as a "quantifiable relationship between the density of an indicator in the water and the potential human health risks involved in the water's recreational use." From such a definition, a criterion can be adopted which establishes upper limits for densities of indicator bacteria in waters that are associated with acceptable health risks for swimmers.

The Environmental Protection Agency, in 1972, initiated a series of studies at marine and fresh water bathing beaches which were designed to determine if swimming in sewage-contaminated marine and fresh water carries a health risk for bathers; and, if so, to what type of illness. Additionally, the EPA wanted to determine which bacterial indicator is best correlated to swimming-associated health effects and if the relationship is strong enough to provide a criterion (EPA 1986: *Ambient Water Quality Criteria for Bacteria - 1986*, EPA 440/5-84-002, U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, DC., NTIS access #: PB 86-158-045).

The quantitative relationships between the rates of swimming-associated health effects and bacterial indicator densities were determined using standard statistical procedures. The data for each summer season were analyzed by comparing the bacteria indicator density for a summer bathing season at each beach with the corresponding swimming-associated gastrointestinal illness rate for the same summer. The swimming-associated illness rate was determined by subtracting the gastrointestinal illness rate in nonswimmers from that for swimmers.

The EPA's evaluation of the bacteriological data indicated that using the fecal coliform indicator group at the maximum geometric mean of 200 organisms per 100 mL, as recommended in *Quality Criteria for Water* would cause an estimated 8 illness per 1,000 swimmers at freshwater beaches.

Newer criteria, using *E. coli* and enterococci bacteria analyses, were developed using these currently accepted illness rates. These bacteria are assumed to be more specifically related to poorly treated human sewage than the fecal coliform bacteria indicator. The equations developed by Dufour (1983: *Health Effects Criteria for Fresh Recreational Waters,* EPA-600/1-84-004, U.S. Environmental Protection Agency, Cincinnati, OH.) were used to calculate new indicator densities corresponding to the accepted gastrointestinal illness rates.

The EPA did not recommend changing the stringency of its bacterial criteria for recreational waters. Such a change did not appear warranted until more information, based on greater experience with the new indicators, can be obtained. The EPA and the State Agencies will then evaluate the impacts of change in terms of beach closures and other restricted uses.

It should be noted that these indicators only relate to gastrointestinal illness, and not other problems associated with waters contaminated with other bacterial or viral pathogens. Common swimming beach problems associated with contamination by nonpoint water pollution sources include skin and ear infections caused by *Psuedomonas aeruginosa* and Shigella.

The Alabama standards for fecal coliforms vary for public water supplies and for the protection of fish and wildlife. The public water supply standards are as follows:

(i) Bacteria of the fecal coliform group shall not exceed a geometric mean of 2,000/100 mL; nor exceed a maximum of 4,000/100 mL in any sample. The geometric mean shall be calculated from no less than five samples collected at a given station over a 30-day period at intervals not less than 24 hours. The membrane filter counting procedure will be preferred, but the multiple tube technique (five-tube) is acceptable.

(ii) For incidental water contact and recreation during June through September, the bacterial quality of water is acceptable when a sanitary survey by the controlling health authorities reveals no source of dangerous pollution and when the geometric mean fecal coliform organism density does not exceed 100/100 mL in coastal waters and 200/100 mL in other waters. When the geometric mean fecal coliform organism density exceeds these levels, the bacterial water quality shall be considered acceptable only if a second detailed sanitary survey and evaluation discloses no significant public health risk in the use of such waters. Waters in the immediate vicinity of discharges of sewage or other wastes likely to contain bacteria harmful to humans, regardless of the degree of treatment afforded these wastes, are not acceptable for swimming or other whole body water-contact sports.

Hardness

The determination of hardness in raw waters subsequently treated and used for domestic water supplies is useful as a parameter to characterize the total dissolved solids present and for calculating chemical dosages for water softening. Because hardness concentrations in water have not been proven to be health related, the final level of hardness to be achieved by water treatment principally is a function of economics. Since water hardness can be removed with treatment by such processes as lime-soda softening and ion exchange systems, a water quality criterion for raw waters used as a public water supply is not given by the EPA.

Nitrates

In quantities normally found in food or feed, nitrates become toxic only under conditions in which they are, or may be, reduced to nitrites. Otherwise, at “reasonable” concentrations, nitrates are rapidly excreted in the urine. High intake of nitrates constitutes a hazard primarily to warmblooded animals under conditions that are favorable to reduction to nitrite. Under certain circumstances, nitrate can be reduced to nitrite in the gastrointestinal tract which then reaches the bloodstream and reacts directly with hemoglobin to produce methemoglobin, consequently impairing oxygen transport.

The reaction of nitrite with hemoglobin can be hazardous in infants under three months of age. Serious and occasionally fatal poisonings in infants have occurred following ingestion of untreated well waters shown to contain nitrate at concentrations greater than 10 mg/L nitrate nitrogen (N) (NAS 1974). High nitrate concentrations frequently are found in shallow farm and rural community wells, often as the result of inadequate protection from barnyard drainage or from septic tanks (USPHS 1961; Stewart, *et al*. 1967). Increased concentrations of nitrates also have been found in streams from farm tile drainage in areas of intense fertilization and farm crop production (Harmeson, *et al.* 1971). Approximately 2,000 cases of infant methemoglobinemia have been reported in Europe and North America since 1945; 7 to 8 percent of the affected infants died (Walton 1951; Sattelmacher 1962). Many infants have drunk water in which the nitrate nitrogen content was greater than 10 mg/L without developing methemoglobinemia. Many public water supplies in the United States contain levels that routinely exceed this amount, but only one U.S. case of infant methemoglobinemia associated with a public water supply has ever been reported (Virgil, *et al.* 1965). The differences in susceptibility to methemoglobinemia are not yet understood, but appear to be related to a combination of factors including nitrate concentration, enteric bacteria, and the lower acidity characteristic of the digestive systems of very young mammals. Methemoglobinemia systems and other toxic effects were observed when high nitrate well waters containing pathogenic bacteria were fed to laboratory mammals (Wolff, *et al.* 1972). Conventional water treatment has no significant effect on nitrate removal from water (NAS 1974).

Because of the potential risk of methemoglobinemia to bottlefed infants, and in view of the absence of substantiated physiological effects at nitrate concentrations below 10 mg/L nitrate nitrogen, this level is the criterion for domestic water supplies. Waters with nitrite nitrogen concentrations over 1 mg/L should not be used for infant feeding. Waters with a significant nitrite concentration usually would be heavily polluted and probably bacteriologically unacceptable.

Dissolved Solids, Conductivity, and Chlorides
Excess dissolved solids are objectionable in drinking water because of possible physiological effects, unpalatable mineral tastes, and higher costs because of corrosion or the necessity for additional treatment.

The physiological effects directly related to dissolved solids include laxative effects principally from sodium sulfate and magnesium sulfate and the adverse effect of sodium on certain patients afflicted with cardiac disease and women with toxemia associated with pregnancy. One study was made using data collected from wells in North Dakota. Results from a questionnaire showed that with wells in which sulfates ranged from 1,000 to 1,500 mg/L, 62 percent of the respondents indicated laxative effects associated with consumption of the water. However, nearly one-quarter of the respondents to the questionnaire reported difficulties when concentrations ranged from 200 to 500 mg/L (Moore 1952). To protect transients to an area, a sulfate level of 250 mg/L should afford reasonable protection from laxative effects.

As indicated, sodium frequently is the principal component of dissolved solids. Persons on restricted sodium diets may have an intake restricted from 500 to 1,000 mg/day (National Research Council 1954). The portion ingested in water must be compensated by reduced levels in food ingested so that the total does not exceed the allowable intake. Using certain assumptions of water intake (*e.g*., 2 liters of water consumed per day) and the sodium content of food, it has been calculated that for very restricted sodium diets, 20 mg/L sodium in water would be the maximum, while for moderately restricted diets, 270 mg/L sodium would be the maximum. Specific sodium levels for entire water supplies have not been recommended by the EPA, but various restricted sodium intakes are recommended because: (1) the general population is not adversely affected by sodium, but various restricted sodium intakes are recommended by physicians for a significant portion of the population, and (2) 270 mg/L of sodium is representative of mineralized waters that may be aesthetically unacceptable, but many domestic water supplies exceed this level. Treatment for removal of sodium in water supplies is also costly (NAS 1974).

A study based on consumer surveys in 29 California water systems was made to measure the taste threshold of dissolved salts in water (Bruvold, *et al*. 1969). Systems were selected to eliminate possible interferences from other taste-causing substances besides dissolved salts. The study revealed that consumers rated waters with 320 to 400 mg/L dissolved solids as “excellent” while those with 1,300 mg/L dissolved solids were “unacceptable.” A “good” rating was registered for dissolved solids less than 650 to 750 mg/L. The 1962 U.S. Public Health Service Drinking Water Standards recommended a maximum dissolved solids concentration of 500 mg/L, unless more suitable supplies were unavailable.

Specific constituents included in the dissolved solids in water may cause mineral tastes at lower concentrations than other constituents. Chloride ions have frequently been cited as having a low taste threshold in water. Data from Ricter and MacLean (1939) on a taste panel of 53 adults indicated that 61 mg/L NaCl was the median level for detecting a difference from distilled water. At a median concentration of 395 mg/L chloride, a salty taste was identified. Lockhart, *et al.* (1955) when evaluating the effect of chlorides on water used for brewing coffee, found threshold taste concentrations for chloride ranging from 210 mg/L to 310 mg/L, depending on the associated cation. These data indicate that a level of 250 mg/L chlorides is a reasonable maximum level to protect consumers of drinking water.

The causation of corrosion and encrustation of metallic surfaces by water containing dissolved solids is well known. By using water with 1,750 mg/L dissolved solids as compared with 250 mg/L, service life was reduced from 70 percent for toilet flushing mechanisms to 30 percent for washing equipment. Such increased corrosion was calculated in 1968 to cost the consumer an additional $0.50 per 1,000 gallons used.

The EPA criteria for chlorides and sulfates in domestic water supplies is 250 mg/L to protect human welfare.

Toxic Organics
Ten of the compounds identified during sampling in the upper Cahaba River reaches have Alabama state standards for the protection of human health, including five compounds that are recognized carcinogens. The following table lists these compounds, and the calculated limits:

	Water and Fish Consumption	Fish Consumption Only
Non-carcinogens:		
2-chlorophenol	0.12 mg/L	0.40 mg/L
Diethyl phthalate	23	118
Dimethyl phthalate	313	2900
Di-n-butly phthalate	3	12
Isophorone	7	490
Carcinogens:		
Benzo(ghi)perylene (PAH)	0.03 μg/L	0.31 μg/L
Benzo(k)fluoranthene (PAH)	0.03	0.31
3,3 Dichloro-benzidine	0.39	0.77
Hexachlorobutadiene	4.5	500
N-nitrosodiphenylamine	50	160

Heavy Metals
Alabama has also established toxic pollutant criteria for human health protection. These criteria are for carcinogens and non-carcinogens and are established for the consumption of both water and fish and for the consumption of fish only. The equations presented by the state of Alabama to calculate these criteria require that a reference dose and a bioconcentration factor be known for mercury and chromium. A cancer potency factor and a bioconcentration factor is also needed for arsenic, a recognized carcinogen. A risk level of 10^{-5} was initially given by the State of Alabama for arsenic causing cancers. This assumes one increased cancer case per 100,000 people associated with this pollutant and fish consumption. The reference doses and bioconcentration factors are now given by the State in their water quality criteria (Chapter 335-6-10, Appendix A). The state removed the arsenic criterion for human health protection in April 1991 and therefore do not include the cancer potency and bioconcentration factors for arsenic. These values are given by the EPA for 10^{-5}, 10^{-6}, and 10^{-7} risk levels (in *Quality Criteria for Water 1986*). The following list shows these criteria for human health criteria protection for fish consumption only:

Arsenic: 0.175 μg/L (calculated using pg. 39, EPA 1986 values for 10^{-5} risk levels)
Chromium(+3): 3433 mg/L (calculated using pg. 95, EPA 1986 and Alabama values)
Mercury: 0.146 μg/L (calculated using pg. 177, EPA 1986 and Alabama values)

Specific numeric criteria have also been established by Alabama for the protection of human health caused by the consumption of fish alone for zinc (5 mg/L). Fish consumption related human health criteria have not been established for cadmium and lead by the State of Alabama.

Historical Observed Concentrations of Pollutants in the Cahaba River (1970-1990)

Upper Cahaba River Water Quality Conditions

Data from 858 samples collected in the upper Cahaba River watershed, except for the Little Cahaba River, were evaluated. This data was evaluated to identify current and likely water quality problems in this area of the Cahaba River watershed. The data was obtained from the EPA's STORET data system and from the Birmingham Water Works Board (BWWB) files and covered the time from 1970 through 1990. The final 858 samples selected for evaluation were derived from more than 1500 samples in the original files. Many samples were eliminated because of mislabeling of the sampling location and for other quality assurance problems. Therefore, the samples evaluated best represent the water quality conditions in the upper Cahaba River.

This report evaluates each water pollution parameter in relationship to existing EPA water quality criteria, Alabama water quality standards, and other information. Of special interest was identifying trends in water quality problems (both in time and location) that could be used to identify water pollution sources.

Typically, different values apply to various beneficial uses of the water bodies. Alabama water uses are listed in the Water Use Classifications for Interstate and Intrastate Waters (ADEM) Code of Alabama Chapter 335-6-11. The Cahaba River is a designated public water supply from the small Highway 280 dam upstream to Grant's Mill Road. All of the watershed also has fish and wildlife as a designated beneficial use. In addition, swimming is a recognized activity in the watershed during the months of June through September. The EPA water quality criteria and the Alabama standards also have separate toxicant criteria for the protection of human health through the consumption of fish and water, or the consumption of fish alone. The fish and water consumption criteria are only applied in areas that are a drinking water supply, while the consumption of fish criteria apply for areas having recreational or commercial fishing activities.

Available water quality data were examined for many upper Cahaba River watershed locations, as described above, in order to demonstrate the geographical extent of any existing water quality problems in the upper reaches of the Cahaba River. It must be realized that the data examined likely do not represent all flow conditions. STORET data generally only contains information from standard long-term monitoring efforts that are mostly obtained during dry weather. Periodic special studies, such as the EPA's Nationwide Urban Runoff Program (NURP) (EPA 1983), may contain wet weather sampling and submit data to a special STORET data file. Unfortunately, STORET generally doesn't contain any information concerning rain history so it is not possible to separate data according to wet or dry conditions. The poor flow information in the STORET files reviewed also did not allow separate wet and dry weather data analyses. It usually rains every three to five days (but only for several hours each time) in the Birmingham area. It therefore only rains for a few percent of the hours of the year. However, rains also will effect the river for some time after the rain stops. The amount of many pollutants occurring in the river during wet weather (maybe affecting about ten percent of the time) can dramatically affect the annual mass loadings of many pollutants. Again, this wet weather data is probably under-represented in the data files available.

Table 6 summarizes the water quality in the upper Cahaba River, as monitored at many locations, as contained in the EPA's STORET and the Birmingham Water Works Board data files for the period of 1970 through 1990.

Table 6. Water Quality Summary for the Upper Cahaba River (1970-1990)

Constituent and units (mg/L, unless noted)	number of obs.	mean	median	stand. dev.	min.	max.
Temperature (^oC)	520	17.5	18	6.8	0.2	31
Turbidity (NTU)	449	14	8.1	19	nd	5
pH	545	7.5	7.5	0.6	5.2	11.9
Hardness (as $CaCO_3$)	237	74	71	31	6.0	194
Dissolved oxygen	486	8.6	8.3	2.1	2.4	14
BOD_5	463	1.4	0.9	1.5	nd	12.4
Total dissolved solids	37	108	104	36	8	212
Total suspended solids	236	7.9	5	9.5	nd	96
Specific conductivity (μmhos/cm)	368	321	178	647	55	5000
Chlorides	102	10	4	20	nd	130
Fecal coliform bacteria (#/100 mL)	184	234	42	1440	nd	19,400
Nitrate nitrogen	232	1.1	0.66	1.3	0.07	9.8
Ammonia nitrogen	69	0.56	0.17	2.4	nd	16
Phosphates (as P)	470	0.27	0.09	0.72	nd	10.3
Arsenic (μg/L)	77	<5	<10	9.6	nd	60
Cadmium (μg /L)	76	<10	<5	22	nd	70
Chromium (μg /L)	178	<7.5	<5	23	nd	138
Copper (μg /L)	150	<0.8	<5	69	nd	530
Iron	456	0.29	0.14	0.59	nd	7.9
Lead (μg /L)	349	25	<1	72	nd	90
Mercury (μg /L)	172	0.32	<0.5	2.5	nd	15
Zinc (μg /L)	151	27	<10	125	nd	870

note: "nd" is not detected. The detection limits vary greatly for each constituent and changed with time as new procedures were used.

The following paragraphs briefly summarize the findings associated with evaluating each water pollution parameter for the upper Cahaba River watershed data.

Temperature

The most critical temperatures for the limited data available for upper Cahaba River fish are 34^{o}C (Largemouth bass - maxima, all times), 32^{o}C (Bluegill, Channel catfish, and largemouth bass - growth, March through November), 27^{o}C (Largemouth bass - embryo survival, October and November), and 18^{o}C (Threadfin shad - spawning, October and November).

More than 520 samples were analyzed for temperature in the upper Cahaba River watershed from 1970 through 1990. The overall range observed was 0.2^{o}C (32^{o}F) to 31^{o}C (88^{o}F), while the average was 17.4^{o}C (64^{o}F). The grouped box plot (Figure 1) shows very little variation in observed temperatures along the watershed, except for a tributary in the upper watershed area that only contained relatively warm samples collected during August of 1987.

Figure 1. Temperature variations in upper Cahaba River.

Only the single sample having the maximum observed temperature exceeded the Alabama 86^{o}F temperature standard for waters having smallmouth bass, sauger, or walleye. Table 7 summarizes the list of fish that were found during an extensive survey of the Cahaba River from 1983 through 1988 (Pierson, *et al*. 1989). Only fish found in the watershed area above the Cahaba River fall-line in Bibb County are included on this list. The 56 fish species on this list are about one-half of the total number of fish species found in the complete watershed. Smallmouth bass, sauger, or walleye are not on this list, so the state standard of 86^{o}F probably would not apply. None of the samples exceeded the 90^{o}F general standard.

Table 7. Fish collected in the Cahaba River During 1983-1988 (Pierson, *et al*. 1989)

Families, species, and common names:

Anguillidae	
Anguilla rostrata	American eel
Clupeidae - herrings	
Alosa chrysochloris	Alabama shad
Dorosoma cepedianum	gizzard shad
Dorosoma petenense	threadfin shad
Cyprinidae - minnows and carps	
Capostoma oligolepis	largescale stoneroller
Ericymba buccata	silverjaw minnow
Hybopsis aestivalis	speckled chub
Hybopsis storeriana	silver chub
Hybopsis winchelli	clear chub
Notropis baileyi	rough shiner
Notropis bellus	pretty shiner
Notropis callistius	Alabama shiner
Notropis chrysocephalus	striped shiner
Notropis stilbius	silverstripe shiner
Notropis texanus	weed shiner
Notropis trichroistius	tricolor shiner
Notropis uranoscopus	skygazer shiner
Notropis venustus	blacktail shiner
Notropis volucellus	mimic shiner
Notropis sp.cf. *longirostris*	
Notropis sp.cf. *volucellus*	

Phenacobius catostomus	riffle minnow
Pimephales virgilax	bullhead minnow
Catostomidae - suckers	
Carpiodes velifer	highfin carpsucker
Erimyzon oblongus	creek chubsucker
Hypentelium etowanum	Alabama hog sucker
Moxostoma duquesnei	black redhorse
Moxostoma erythrurum	golden redhorse
Ictaluridae - freshwater catfishes	
Ictalurus punctatus	channel catfish
Noturus leptacanthus	speckled madtom
Noturus munitus	frecklebelly madtom
Cyprinodontidae - killifishes	
Fundulus olivaceus	blackspotted topminnow
Poeciliidae - livebearers	
Gambusia affinis	mosquitofish
Centrarchidae - sunfishes	
Ambloplites ariommus	shadow bass
Lepomis cyanellus	green sunfish
Lepomis macrochirus	bluegill
Lepomis megalotis	longear sunfish
Lepomis microlophus	redear sunfish
Lepomis punctatus	spotted sunfish
Micropterus coosae	redeye sunfish
Micropterus salmoides	largemouth bass
Percidae - perches	
Ammocrypta beani	naked sand darter
Ammocrypta meridiana	southern sand darter
Etheostoma histrio	harlequin darter
Etheostoma jordani	greenbreast darter
Etheostoma rupestre	rock darter
Etheostoma stigmaeum	speckled darter
Etheostoma (Ulocentra) sp.	
Percina aurolineata	goldline darter
Percina lenticula	freckled darter
Percina nigrofasciata	blackbanded darter
Percina shumardi	river darter
Percina vigil	saddleback darter
Percina sp. cf. *caprodes*	
Percina sp. cf. *copelandi*	
Cottidae - sculpins	
Cottus carolinae	banded sculpin

The only EPA criteria that were exceeded were the spawning season criteria, applicable for October and November. 82 of the 521 samples were collected during these two months and 25, or 30 percent, of the samples exceeded the critical 18^{o}C criteria for threadfin shad. These were generally evenly spread over the length of the river sections, except none of the temperatures from the BWWB pump station location exceeded this spawning season criteria. Unfortunately, EPA critical temperatures were only available for three to four of the 56 fish species likely to be in the river section of interest. It is possible that some of the other fish present would have more restrictive temperature criteria than the treadfin shad.

None of the temperature values exceeded the state general 90^{o}F standard, and only one of 521 values exceeded the smallmouth bass 86^{o}F State of Alabama standard. However, about one-third of the temperatures obtained during the October and November spawning season exceeded the EPA critical value for threadfin shad. Other fish may also be adversely affected during this critical time period, but temperature criteria are not available from EPA. Therefore, it is assumed that temperature is generally not of major concern, especially considering the high natural water temperatures in the area, except during critical spawning times. Care must be taken not to raise water temperatures by removing stream shading or by careless detention pond use that can significantly raise water temperatures.

pH

The EPA recommended water quality criteria for pH restricts pH values to be in the range of 5 to 9 for domestic water supplies (welfare), and within the range of 6.5 to 9.0 for freshwater aquatic life protection. The State of Alabama's fresh water pH standards for public water supplies and aquatic life are: “Sewage, industrial wastes or other wastes shall not cause the pH to deviate more than one unit from the normal or natural pH, nor be less than 6.0, nor greater than 8.5.”

Almost 550 pH observations were made in the upper Cahaba River watershed from 1970 through 1990, as shown in the grouped box plots in Figure 2. The overall observed pH range was 5.2 to 11.9, and the average pH value was 7.5. The values were compared to the above listed EPA criteria and Alabama standards. Few pH problems were noted, except for an unnamed tributary located about 2.1 miles upstream of the BWWB pumping station. Six of the eight samples exceeded all of the standards due to their high pH values (9.7 to 11.9). This tributary is likely affected by an old landfill.

The average pH values were very similar for all river reaches, but the ranges increased in an upstream direction, excepting the above mentioned tributary. Almost all of the samples were in the range of 6.5 to 8.5 and therefore within the range of the criteria and standards. However, about two percent of the pH observations were in violation of the standards. About half of the violations were associated with the high pH values from the tributary, but most other violations were because of pH values that were too low. The low values (between pH 5 and 6) were mostly from a location about 28 miles upstream of the pumping station that had a lot of sampling activity. Except for the local source problem in the tributary, pH does not appear to be a major problem in the upper watershed.

Figure 2. pH variations in upper Cahaba River.

Hardness

Between 1970 and 1990, 238 samples were analyzed for hardness from the upper Cahaba River watershed. The hardness samples obtained at the BWWB pump station were all collected during the 1970s. Most of the hardness samples were collected 28 miles upstream of the pump station from 1974 through 1987. The range of observed hardness was from 6 to 194 mg/L, as $CaCO_3$, and the average hardness was 74 mg/L, as $CaCO_3$. The highest hardness values appear to be generally in the most upstream area of the watershed, including the tributaries. This is expected, considering that groundwater is harder than surface waters, and that these upstream waters are more affected by groundwaters than by rain runoff. Several abandoned mines also exist in the upper reaches of the watershed and mine runoff is also usually quite hard.

More than half of the samples were soft (<75 mg/L, as $CaCO_3$), and slightly less than half were moderately hard (between 75 and 150, mg/L as $CaCO_3$). Only 3 percent of the samples were hard (150 to 300, mg/L as $CaCO_3$). The predominantly soft water in the Cahaba River results in more restrictive heavy metal criteria and standards. Unfortunately, few of the samples analyzed for heavy metals in the upper watershed area were also analyzed for hardness and alkalinity, making it difficult to directly evaluate the metal analyses for the hardness defined water quality criteria.

Dissolved Oxygen and Biochemical Oxygen Demand

Almost 500 DO and BOD analyses were made on samples collected from the upper Cahaba River watershed between 1970 and 1990. Most of the analyses (about 300) were from the uppermost portion of the watershed, above 24 miles upstream from the BWWB pumping station. A series of 16 monthly samples were obtained from many locations during 1983 and 1984 throughout the watershed. The observed DO values ranged from 2.4 to 14 (with a mean of 8.6), while the observed BOD_5 values ranged from <1 to 12.4 mg/L. The following table summarizes the percentage of samples having DO values less than various critical values, by watershed location:

		Percentage of samples less than DO conc. (mg/L):							
Area	**Number of Samples**	**<3**	**<4**	**<5**	**<5.5**	**<6**	**<6.5**	**<8**	**<9.5**
1	9	0 %	0 %	0 %	0 %	0 %	22 %	56 %	89 %
2	48	0	0	0	0	0	2	35	63
3	48	0	0	2	6	13	15	42	65
4	24	0	0	13	17	38	46	71	79
5	358	0.3	1	4	6	11	16	42	61
all	487	0.2	1	4	6	11	16	43	63

- Area 1 is the BWWB pump station (mile 0),
- Area 2 is between 1 and 14 miles upstream of the pump station, and includes 3 unnamed tributaries,
- Area 3 is between 14 and 18 miles upstream of the pump station, and includes 2 unnamed tributaries,
- Area 4 is between 18 and 23.8 miles upstream of the pump station, and includes 1 unnamed tributary, plus Big Black Creek, and
- Area 5 is between 23.8 and 35 miles upstream of the pump station, and includes Pinchgut Creek, and the north and south branches of Little Cahaba Creek.

The grouped box plots in Figure 3 show that the DO had similar median values in all of these river reaches, but the overall range of values (minimums and maximums) increased for further upstream distances. The most consistently low DO values were seen to be in tributary 500 (north branch of Little Cahaba Creek). The lowest DO (2.4 mg/L) was observed in the south branch of Little Cahaba Creek on October 31, 1984. This very low DO would produce acute mortality for all life stages of many fish and invertebrates. Other DO values (obtained between June 1983 and October 1984) from this same location were all greater than 7.7 mg/L.

Figure 3. DO variations in upper Cahaba River.

About 4 percent of all DO values were less than the State of Alabama standard of 5 mg/L, a value that would produce moderate impairment for early life stages of nonsalmonid fish and slight impairment for other fish life stages and invertebrates. One percent of the samples had DO values less than 4 mg/L, the absolute minimum standard for Alabama, and would cause acute mortality of early life stages of nonsalmonid fish and moderate impairment of other life forms. Acute mortality of invertebrates would also occur at DO values lower than 4 mg/L. The EPA criteria for warm water (nonsalmonid) is 6.5 mg/L if no impairment of fish were to occur. 16 percent of the samples had DO values less than this value. In addition, the no impairment DO criteria to protect invertebrates is 8 mg/L and was exceeded by about 40 percent of the samples.

Almost all of the low DO values occurred during critical summer months when the stream temperatures were highest. The scatterplot of DO and temperature (Figure 4) shows a very significant trend of low DO with high temperatures. Most of the DO values lower than 5 mg/L occurred with high temperatures (between 25 and 30^{o}C) which would further add to the organism's stress.

Figure 4. Scatterplot of DO and temperature in the upper Cahaba River.

These data indicate that the upper Cahaba River has experienced periodic problems associated with DO (less than 4% of the samples were in violation of the Alabama 5 mg/L standard). However, the infrequent observations at most locations (generally monthly, at best) may shield some of the more serious problems. In eutrophic waters, the lowest DO values would occur during late night and very early morning hours, as an example. Observations during periods of strong sunlight at these same locations would show high DO values. These data indicate that DO is not expected to be a persistent problem in these waters.

Turbidity and Suspended Solids

Almost 450 turbidity and 250 suspended solids analyses were conducted on Upper Cahaba River watershed samples from 1974 to 1990. The median turbidity value observed was a low 8 NTU, while the highest value observed was 185 NTU. 25 of the 449 turbidity values were greater than 50 NTU. The frequency of Alabama criteria violations (an increase of less than 50 NTU above "background"), based on these data, would therefore be less than six percent. The median suspended solids value was 5 mg/L and the maximum was 96 mg/L. Figure 5 is a grouped box plot showing the variation of turbidity along the upper Cahaba River. The highest turbidity values were from samples obtained from the uppermost stretch of the Cahaba River, above about 24 miles upstream of the Birmingham Water Works Board's pumping station. It is known that suspended solids and turbidity values obtained during, or soon after, rains would be much higher than after extended dry periods.

Figure 5. Turbidity in the upper Cahaba River.

Suspended solids and turbidity are mostly caused by erosion of soils during rains. Natural erosion in the Birmingham area is high because of the erodible soils, steep slopes, and high energy rains. However, erosion caused from disturbances (such as construction activities, farming or forestry operations) also causes very high suspended solids and turbidity discharges. The construction site erosion rate in the Birmingham area has been measured to be from 150 to more than 300 tons per acre per year, for example. These rates are about ten times the national average for construction sites. Without adequate erosion controls, these very high discharges cause substantial downstream damage, including excessive exceedences of the above stated criteria. The actual frequency of exceedences of these criteria, are therefore expected to be much greater than shown with this data. Especially of concern is the minimum Alabama criterion of prohibiting sediment deposition that interferes with beneficial uses. Many urban streams experience excessive sedimentation that is expected to have been at least partially responsible for the dramatic decline in fish and other aquatic life that is found in urban creeks.

Total Dissolved Solids, Chlorides, and Conductivity

Only 38 TDS measurements were made in the upper Cahaba River study area during the period of 1970 through 1990. However, 368 conductivity and 103 chloride measurements were taken during this period. The limited TDS data had a range of 58 to 212 mg/L, and the chloride range was from not detected to 130 mg/L. The more numerous conductivity measurements ranged from 55 to 5000 μmhos/cm. The chloride values were all less than the criteria of 250 mg/L for domestic drinking water and less than the Alabama Cahaba River standard of 250 mg/L for fish and aquatic life. However, the high conductivity measurements indicate that chloride concentrations were very likely to exceed the criteria and standard values.

The highest conductivity values were observed along the main Cahaba River channel, and in Black Creek. There was apparently a relatively long-term discharge of saline water in Black Creek above 4.8 miles upstream of the confluence of Black Creek with the Cahaba River in July of 1990. The conductivity measurements in Black Creek were between 3200 and 5000 μmhos/cm on two sampling days 15 days apart. Conductivity measurements in the Cahaba River on these same sampling days were: 2600 and 2800 μmhos/cm at 26.6 miles upstream of the BWWB pump station, 2600 and 2900 μmhos/cm at 16.1 miles upstream of the BWWB pump station, 2700 μmhos/cm at 11.4 miles upstream of the BWWB pump station, and 3900 μmhos/cm at the pump station. High conductivity measurements (900 to 2600 μmhos/cm) were also observed 0.3 miles up a tributary at 2 miles upstream of the pump station from 1981 through 1984 (when the last samples were obtained). This sampling station was downstream of a landfill in Mt. Brook.

Analyses were also made to compare TDS, conductivity and chloride relationships. A scatterplot (Figure 6) show a reasonably good relationship between TDS and conductivity, with the TDS values (in mg/L) about one-half of the conductivity values (in µmhos/cm). All of these pairs of data were obtained at the BWWB pump station sampling site from 1973 through 1979. It is expected that the relationship would be different for other locations and times. The TDS concentrations were also about 30 times the chloride concentrations simultaneously observed. ADEM has allowed chloride dischargers to use a relationship between chlorides and conductivity to enable discharge permit reporting based on simpler conductivity measurements. Unfortunately, the scatterplot showed a very poor relationship between chlorides and conductivity for the Cahaba River system as a whole. Even at a single location, the ratio of conductivity to chloride measurements varied over a wide range (from 2 to 2300), making the use of this ratio to predict chloride measurements in receiving waters affected by wastewaters not very useful. A similar scatterplot of suspended solids and turbidity was showed a large amount of scatter, with little use.

The dissolved minerals in the Cahaba River probably do not exceed any of the chloride standards or criteria during normal conditions. However, intermittent discharges of saline waters from upstream sources have been shown to have dramatic effects for large distances downstream of the discharge. The chloride concentrations at many locations in the river are likely to exceed the criteria and standards during these periodic discharges.

Fecal Coliform Bacteria

Fecal coliform bacteria were measured in 185 samples in the area of interest. The median value observed was 42 organisms per 100 mL, but the highest value was almost 20,000 per 100 mL. This very high observation was from a single sample obtained at the BWWB pump station in 1977. The next highest observations were six samples that had fecal coliform counts between 1000 and 1500 organisms per 100 mL. These high values were from different locations and times along the Cahaba River. Several of the small tributaries had much lower fecal coliform counts (<100 organisms per 100 mL) than along the main reaches of the Cahaba River. 65 of the 185 analyses were for samples collected during June through September, months when the Alabama swimming criteria apply. 15 of these samples (23 percent) of these samples exceeded the 200 organism per 100 mL swimming criteria.

Figure 6. Scatterplot relating conductivity and TDS in samples from the upper Cahaba River.

It is not uncommon for urban runoff to have fecal coliform counts of between 10,000 and 100,000 organisms per 100 mL which could contaminate large amounts of receiving waters. However, fecal coliforms in urban runoff are probably a poor indicator of gastrointestinal disease. Unfortunately, pathogens that cause skin and ear infections can be very common in urban runoff. Sampling close to runoff events will likely have much greater bacterial densities than after long dry periods. Discharges of poorly treated sanitary sewage and SSOs (sanitary sewer overflows) are also known to have occurred in the Cahaba River during this monitoring period. Bacteria potentially affecting water contact recreation are likely a problem in the Cahaba River area, especially considering the pathogens that are not well indicated by fecal coliforms.

Nitrate Nitrogen

Nitrate nitrogen concentrations were obtained from 233 samples from the upper Cahaba River area from 1970 through 1990. The maximum concentration observed was 9.8 mg/L, while the median concentration observed was about 0.7 mg/L. Therefore, all nitrate concentrations were below the 10 mg/L critical value for a public water supply, although five samples had concentrations greater than 5 mg/L. The nitrate concentrations do not appear to vary greatly with time or location, based on these data, as shown in Figure 7. With increased ammonia discharges into the river, probably associated with urbanization, the frequency of high nitrate concentrations will increase, with some eventual criteria exceedences.

Ammonia and Kjeldahl Nitrogen

The observed total ammonia concentrations from Cahaba River watershed samples ranged from <0.2 mg/L to 16 mg/L. The median concentration was 0.17 mg/L. As shown on the attached grouped box plot (Figure 8), all 69 observations reported (from March 1977 to August 1987) were less than 1 mg/L, except for two values that were very high (13.1 and 16 mg/L). These two high values were observed in a tributary (upper Little Cahaba River, about 28 miles above the Cahaba River pump station, between Big Black Creek and Pinchgut Creek). A series of samples were taken along this tributary on August 11, 12, and 13, 1987 to investigate high ammonia concentrations,

Figure 7. Nitrate variations in the upper Cahaba River.

Figure 8. Ammonia variations in the upper Cahaba River.

apparently associated with a chicken manure discharge problem. The data indicate that these high ammonia concentrations were only present very close to the most upstream site sampled. Samples taken downstream on these days were all less than 1 mg/L ammonia. High ammonia concentrations were present on both days that this upstream site was sampled.

The ammonia data and criteria summary shows that the applicable ammonia criteria (based on concurrent pH and temperature conditions) were only exceeded twice (for the two highest concentrations, discussed above). These violations were very large (12.6 and 15.2 times the criteria) and existed for at least two days and therefore could be expected to have caused a severe water pollution incident (though limited in area). However, most of the observed concentrations were between 0.1 and 0.6 of the criteria.

Most of the river system is nitrogen limited and dominated by point sources (sanitary sewage discharges). Current ammonia limitations on treated sewage discharges consider this problem, but additional ammonia loadings are inevitable with increased urban runoff. Any additional nitrogen discharges could significantly worsen the critical nutrient enrichment (eutrophication) conditions in the river.

Phosphate

Concentrations of phosphate forms of phosphorus were obtained from 471 samples, more than for any other analyses, except for temperature and pH. Most of the observed concentrations were between 0.05 and 1 mg/L, with several near or larger than 5 mg/L. All locations had similar concentrations, except for the North Fork of the Little Cahaba Creek (about 28 miles above the BWWB pump station). The concentrations in this creek reach were much greater than elsewhere, ranging from 1 to 10 mg/L. These samples were all obtained on August 11 through 13, 1987. These same samples also had the highest ammonia concentrations observed and were obtained during the investigation of an apparent excessive chicken manure discharge. Therefore, the typical range of total phosphate in the upper Cahaba River system is 0.05 to 1 mg/L (as P).

Almost all of the phosphate observations from the main reaches of the Cahaba River were excessive when compared to the EPA recommended values to prevent eutrophication. Most of the tributary phosphate concentration values were less (with the exception noted above). The highest concentrations were generally found below about mile 28 from the pump station. Because of the persistence (in time) and wide spread nature of these relatively high concentrations, it is expected that eutrophication is a significant threat to the Cahaba River water quality, especially during summer months when the flows are less and in the presence of sunlight.

Eutrophication is dependent on the excessive presence of all required nutrients. Algae requires phosphorus, nitrogen and carbon, along with many minor nutrients that are not generally limited. Carbon is also plentiful through organic matter (both natural and man-caused) and is rarely a limiting factor. Therefore, most determinations of eutrophication require simultaneous analyses of available nitrogen and phosphorus compounds. If the N/P ratio is less than 10, the river is assumed to be nitrogen limited (relatively rare in U.S. waters) and is point source dominated (especially by sanitary sewage discharges). If the N/P ratio is more than 10, the river is assumed to be phosphorus limited (much more common) and is nonpoint source dominated (such as by agriculture and urban runoff).

Ammonia nitrogen (as N) and phosphates (as P) were simultaneously monitored in 16 Cahaba River samples at 21 to 32 miles above the pump station, and in 10 tributary samples. All of the simultaneous ammonia and phosphate observations were obtained during August 12 through 14, 1987, apparently as part of the chicken manure discharge investigation. During this short investigation, the N/P ratio was 10, or greater, in Black Creek and in another tributary slightly downstream of Black Creek. This implies that most of the river system was nitrogen limited and dominated by point sources, except for the local area noted.

The average ammonia nitrogen concentration is about 0.15 mg/L and the phosphate concentration is about 0.1 mg/L over a long period of time in the main Cahaba River near the pump station. The resulting N/P ratio is expected to be about 1.5, signifying nitrogen limiting conditions and significant influences from sanitary sewage discharges. Ammonia nitrogen concentrations of 0.15 mg/L could result in chlorophyll concentrations of about 15 ug/L, which is greater than the commonly accepted value of 10 ug/L for eutrophic conditions (if a lake). The 0.1 mg/L phosphate concentration could produce chlorophyll concentrations of 100 ug/L, which would be ten times the eutrophic limit.

Therefore, any additional nitrogen discharges could significantly worsen the currently marginal conditions in the river and in downstream waters. Future nitrogen discharges are likely from increased nonpoint water discharges, especially associated with landscaping fertilizers. As noted previously, this is an unusual condition, as most waters are phosphorus limited, leading to severe restrictions on the use of phosphorus fertilizers and detergents. The Cahaba River received more sanitary sewage discharges than would be expected based on the amount of urban development in its' watershed during this monitoring period. Ammonia limitations on the treated sewage discharges consider this problem, but additional ammonia loadings are inevitable with increased urban runoff.

Arsenic
The 78 reported filtered arsenic concentrations observed above the Cahaba River pump station operated by the BWWB ranged from <10 μg/L to a high of 60 μg /L. Most of the reported concentrations were below the detection limit (mostly 10 μg /L) and none of the observed arsenic concentrations exceeded the fish and wildlife criteria (190 μg /L was the lowest aquatic life criterion). However, many of the observations exceeded the EPA human health criteria. Only one sample (obtained at the BWWB pump station in 1979) exceeded the revised Alabama standard to protect human health. Unfortunately, the EPA human health standards were much less than the detection limits. Therefore, observed arsenic concentrations reported to be less than the detection limits do not imply that the EPA criteria were not exceeded. In fact, all of the reported concentrations greatly exceeded the EPA criteria, by 5 to more than 2500 times.

Most of the arsenic observations were obtained at the pump station. Arsenic was also measured at 16, 28, and 32 miles upstream of the pump station, but only one of these 29 upper river samples had detectable concentrations. It is likely that arsenic concentrations are between 1 and 10 μg /L over much of the watershed study area.

Based on the EPA guidance, these data imply that a significant increase in cancer may be associated with arsenic in the public drinking water supply. These criteria assume standard water treatment. Additional treatment to remove arsenic is not being used by the BWWB. Because of the linear relationship assumed between arsenic concentrations and increased cancer incidence, a 45 times exceedence of the 10^{-5} risk standard (associated with a total arsenic concentration of 1 μg/L) results in a $4.5X10^{-4}$ risk. Similarly, a 450 times exceedence of the 10^{-5} risk standard (associated with a total arsenic concentration of 10 μg/L) results in a $4.5X10^{-3}$ risk. With a million people being served by this water supply, 450 to 4,500 additional cancer cases may occur over each generation having a lifetime exposure drinking this water and eating fish from the river. If only fish are eaten, and the water is not consumed, then the increased cancer incidence would decrease to about 60 to 600. Therefore, most of the risk is associated with water consumption, by far the most common exposure route for arsenic.

Cadmium
Only 7 of the 77 cadmium observations from 1970 to 1990 in the upper Cahaba River area had detectable concentrations. The detection limits varied from 1 to 100 μg/L, while the observed concentrations ranged from 1 to 70 μg/L. It is difficult to compare the criteria with the observations, because most of the applicable cadmium criteria were much less than the detection limits. The observed values occurred in 1970, 1972, 1973, 1979, and 1990, at the pumping station and in Black Creek.

The available criteria for freshwater aquatic life is dependent on concurrent hardness concentrations. About 1/3 of the cadmium observations did not have associated hardness values, and aquatic life criteria could not be calculated for these values. Chronic freshwater cadmium standards ranged from 0.24 µg/L (associated with a hardness value of only 14 mg/L as $CaCO_3$) to 1.33 µg/L (associated with the maximum hardness value of 125 mg/L as $CaCO_3$). Most all of these criteria were therefore much less than the detection limits of the analyses used. Two of the three detectable cadmium concentrations that had hardness values exceeded their associated chronic criteria by 1.8 and 107 times.

Similar problems with detection limits and the lack of hardness values affected the acute criteria comparisons. One of the three available values exceeded the associated criteria (by 38 times). All of the cadmium concentrations that exceeded the aquatic life criteria were from the pumping station location.

The human health criterion associated with public water supplies (10 µg/L) is not dependent on hardness. Four of the seven detectable cadmium concentrations exceeded this human health criterion. The exceedences ranged from 1.2 to 7 times the criterion. Two of these exceedences were observed at the pumping station (a public water supply) and two were observed in Black Creek (a tributary to a public water supply, even though it does not have a public water supply designated use).

It is expected that most of the cadmium concentrations would have exceeded the chronic aquatic life criteria, if the detection limits were appropriate and if complete hardness data were available. Many concentrations would probably have also exceeded the acute aquatic life criteria and the human health criterion, for similar reasons.

Chromium
Chromium data were obtained from 179 samples from the upper Cahaba River area. Most of the observations were below the detection limits of the analyses procedures used (generally from 2 to 10 µg/L). These samples were obtained from all areas of the study area and from 1970 through 1990. The maximum recorded concentrations were all found in the upper reaches of the Cahaba River (above 28 miles from the BWWB pumping station). The chromium concentrations in this reach were from 25 to 138 µg/L. The chromium concentrations downstream from this reach and from the tributaries were all 10 µg/L, or less.

T he chromium observations were for filterable portions of the metal, but were not distinguished as to their valence state. The Cr^{+6} state is much more toxic than Cr^{+3}, but is usually found in much smaller quantities. It is assumed that almost all of the chromium detected was as Cr^{+3}. The EPA freshwater aquatic life criteria and Alabama standard for Cr^{+3} are dependent on concurrent water hardness values. Most of the chromium observations do not have associated hardness values. However, it is unlikely that Cr^{+3} causes an aquatic life problem, even at the highest recorded concentration of 138 µg/L. The lowest Cr^{+3} standard, associated with the lowest observed hardness value, was 41 µg/L which would be very rare. Typical 4-day chronic Cr^{+3} standards are between 100 and 200 µg/L. Only two of the 179 samples were 100 µg/L, or greater. Most of the detection limits were much less than these criteria and standards and were appropriate for these analyses. Acute Cr^{+3} standards were much greater, being about 1500 to 2500 µg/L, much greater than any chromium observations.

The EPA's Cr^{+6} aquatic life criteria are much more critical than for Cr^{+3}. These are 16 µg/L for the acute 1-hr criterion and 11 µg/L for the chronic 4-day criterion. If all of the observed chromium was Cr^{+6} (highly unlikely), then these criteria would be exceeded by 1.6 to 12 times for three observations (out of 179), all located in the upper reach of the Cahaba River.

The EPA's human health criterion for Cr^{+3} was 3433 µg/L, which is much greater than any observed chromium concentration. The Cr^{+6} EPA human health criterion was 50 µg/L which may be periodically exceeded in the upper reach of the Cahaba River, if the chromium observations were all Cr^{+6} (very unlikely) and if the reach had a designated use as a public water supply (even though it is a tributary to one). It is therefore unlikely that chromium causes an aquatic life or human health problem in these waters.

Copper
Copper observations were well distributed throughout the upper Cahaba River area and from 1970 through 1990.The maximum copper concentration observed in the 151 samples available for the Cahaba River study area was 530 µg/L. Like chromium, the largest observed concentrations were from the upper reach of the Cahaba River, at about 28 miles upstream of the BWWB pump station. The highest concentrations were found at this area during 1974 through 1976. Copper concentrations since that time have generally been below the detection limits (an unfortunate high detection limit of 50 to 100 µg/L, due to the older available equipment). Other copper observations at lower reaches of the Cahaba River, near the pump station, ranged from 2 to

15 µg/L during this mid 1970s time period. Since 1983, the detection limits have been substantially reduced to about 5 µg/L and more recent copper observations have been less than this limit. No copper observations from tributary samples (all from 1983 and 1984) were greater than the 5 µg/L detection limit.

Again, most of the samples were not analyzed for hardness, making it impossible to calculate an appropriate standard for many of the observations. The most critical aquatic life standards occur during low hardness conditions. The lowest hardness recorded was 23 mg/L as $CaCO_3$, with an associated chronic 4-day copper criterion of 3.37 µg/L. The acute 1-hr copper criterion for this critical hardness condition was 4.44 µg/L. Most of the chronic standards calculated were in the range of 5 to 14 µg/L and the acute standards in the range of 8 to 20 µg/L. The last copper observations that exceeded any of these standards and criteria were from 1977. Two copper values of 14 and 50 µg/L (but without concurrent hardness values) were obtained in 1983 and 1984, indicating the potential for infrequent standard violations.

The human health criterion of 1,000 ug/L copper was only approached by the series of older samples previously described. These highest copper observations were still only 10 to 50 percent of this criterion. The largest observed concentrations of copper were found in the upper reach of the Cahaba River. The human health criterion is not likely exceeded in the area of the watershed studied. However, infrequent violations of the EPA aquatic life criteria may occur.

Iron

Iron was frequently monitored (457 observations were reported) in the upper Cahaba River area during the period from 1970 to 1990, probably because of its importance in areas having coal mining activity. The maximum iron concentration was 7.9 mg/L, while the median concentration was 0.14 mg/L. Seven high values were greater than 2 mg/L. As shown on the grouped box plot (Figure 9), the highest iron concentrations were found at the most upstream reach of the Cahaba River and the highest values at the different locations decreased in the downstream direction. The highest concentration of iron recorded was from a tributary (Pinchgut Creek) and was obtained in 1983. Other iron analyses in Pinchgut Creek (only sampled in 1983 and 1984) were all 0.12 to 1.35 mg/L (a typical range for iron elsewhere).

The EPA aquatic life 1-hour acute criterion of 1.0 mg/L was exceeded infrequently in the Cahaba River, from the pumping station to about 28 miles upstream. Five of 279 reported samples exceeded the criterion in this downstream reach during the period 1970 through 1990. The exceedences were less than two times the standard in this river reach. The frequency and magnitude of exceedences increased in the upper river reach, to about four times the criterion. Fifteen of 65 samples exceeded this criterion in the upstream reach, implying frequent criterion violations that would be greater than the once per three years suggested maximum exceedence frequency for aquatic life criterion.

The EPA human health (welfare) criterion was more frequently exceeded in all samples, compared to the aquatic life standard. In the lower reaches of the Cahaba River (from the pumping station to 28 miles upstream), the iron concentrations were up to four times the criterion and 48 of 279 samples exceeded the criterion. At the 28 mile location, three samples had iron concentrations more than ten times the human health (welfare) criterion. In the upper river reach, 37 of the 65 samples exceeded this criterion. These upper reaches do not have a designated use as a public water supply, but they are tributaries to the public water supply. The EPA aquatic life 1-hour acute criterion was exceeded infrequently in the Cahaba River. However, the EPA human health (welfare) criterion was frequently exceeded. There are no Alabama state standards for iron.

Lead

More samples (350) were analyzed for lead in the upper Cahaba River area than for any other heavy metal during the 1970 to 1990 period. The maximum observed concentration was 90 µg/L, but many of the samples had lead concentrations below the detection limits (usually 10 or 50 µg/L). As with most of the metals, the highest concentrations were associated with samples obtained from the upper reach of the Cahaba River (at 25 and 34 miles

Figure 9. Iron variations in upper Cahaba River.

above the BWWB pumping station). However, lead concentrations greater than 10 μg/L were detected at many locations along the river (from 4 miles above the pumping station, and above) and for current samples. Most of the lead observations were obtained during 1988, but earlier years (since 1972) and 1990 are also represented. Only Black Creek of the tributaries had detectable lead concentrations. However, the detection limit for all of the other tributary samples was a high 50 μg/L.

The EPA aquatic life criteria and the Alabama standards for lead are dependent on hardness. Unfortunately, many of the lead analyses did not have concurrent hardness observations, making it impossible to evaluate all of the lead observations for aquatic life problems. The most critical lead standards occur during low hardness values. The lowest hardness value observed with the lead analyses was 24 mg/L as $CaCO_3$. The associated chronic 4-day standard for this hardness level is 0.52 μg/L while the acute 1-hr standard is 13.3 μg/L. Most of the chronic 4-day standards calculated were in the range of 1 to 4 μg/L and the acute 1-hr standards were in the range of 20 to 70 μg/L. The newer samples had detection limits that were less than 1 μg/L, making appropriate comparisons possible. However, the earlier samples had much greater detection limits of about 10 μg/L, or periodically greater. Few samples, especially newer samples, had both the necessary low detection limits and concurrent hardness values to make adequate comparisons with the standards. However, many of the lead observations were in the range of the standards. It is expected that the majority of lead concentrations would exceed the chronic 4-day lead standards over much of the study area, by probably less than ten times the standards. It is expected that the acute 1-hr standards would be rarely exceeded.

The human health standard of 50 μg/L was only closely approached or exceeded three times (out of 350). These high values were 48 and 90, (in 1988 and 1984 at 25 miles above the pumping station) and 50 μg/L (at 34 miles above the pumping station in 1983). These sampling locations were not in the public water supply portion of the study area, but are upstream tributaries to the public water supply.

Mercury

The highest mercury concentration reported from the 173 samples obtained from the upper Cahaba River area during the period from 1970 to 1990 was 15 μg/L. About ten other samples had mercury concentrations greater than 5 μg/L. Many of these high mercury observations were obtained from samples from the upper reach of the Cahaba River, as shown in Figure 10, but some were also from two unnamed tributaries located about 10 miles upstream of the BWWB pumping station. Few reported mercury analyses have been conducted since 1984, except for a relatively continuous series of mercury analyses from a location 28 miles upstream of the pump station.

Figure 10. Mercury variations in upper Cahaba River.

The detection limit for the mercury analyses was generally 1 μg/L, substantially greater than the chronic aquatic life standard and criterion of 0.012 μg/L and greater than the EPA human health criteria of 0.144 and 0.146 μg/L. Therefore, an undetected mercury concentration does not mean that the sample did not reflect deleterious conditions. The 1 μg/L detection limit was less than the acute aquatic life standard of 2.4 μg/L.

All of the detected mercury observations (about half of the samples) greatly exceeded the chronic aquatic life standard. Typical exceedences were 50 to several hundred times the standard. Fifteen of the 173 sample analyses also exceeded the acute aquatic life standard, but by smaller amounts (by 1 to 5 times). All of the detected mercury observations also exceeded the human health criteria, by up to 100 times.

Zinc

The highest zinc concentration observed in the 151 samples obtained from the upper Cahaba River area during the period from 1970 to 1990 was 870 μg/L. The most recent zinc data reported was obtained in 1984. Probability analyses indicated that three samples exceeded 500 μg/L. Two of these (800 and 870 μg/L) were located in the upper Cahaba River reach at 28 miles above the BWWB pumping station. These high observations were obtained in 1974 and 1975. More recent zinc observations in that area were all less than 100 μg/L. Two high samples (530 and 430 μg/L) were obtained at the pumping station in 1977 and 1978. No reported zinc data are available for the pump station since 1979. Most zinc concentrations are in the range of 10 to 100 μg/L. This is also the range of the detection limits used, so there are a large number of samples having concentrations below the detection limits.

Zinc criteria and standards are also dependent on concurrent hardness values. Only a few of the sampling locations had hardness observations (the pump station from 1970 through 1979, 28 miles upstream of the pump station for 1974 through 1977, and for a single analysis at 32 miles upstream of the pump station in 1980. This lack of associated hardness data makes it difficult to compare the zinc observations with the Alabama aquatic life standard and the EPA “never to be exceeded” aquatic life criteria. The EPA 1-day criteria is not associated with hardness (47 μg/L) and is seen to have been frequently exceeded by up to 10 times, depending on location. The other aquatic life criteria and standards were periodically exceeded, but the data are quite old and probably are not indicative of current conditions.

The EPA's human health criteria (5,000 μg/L) was never approached. As noted above, the highest zinc concentrations reported were less than 1,000 μg/L.

Organic Toxicants

No data were available for organic priority pollutants in the data bases or files investigated for this report. Therefore, six samples were collected at various Cahaba River locations in March of 1991 and analyzed for organic priority pollutants. The locations sampled included: Riverrun, Grant's Mill Rd., Riverview Rd., County 10 (CVCC), Trussville Park, and at Old Springville Rd. These locations were all along the main channel of the Cahaba River. The organics having the highest concentrations were found at all sampling locations, while those only found at trace concentrations were generally only found at one or two locations. The phthalate esters were the most abundant and common organic pollutants found. These are plasticizers and are commonly found in runoff from urban areas, from sewage treatment plants, and from specific industries.

None of the non-carcinogens are likely present in quantities in excess of the Alabama standards. However, the carcinogens have much more stringent limits, especially the PAHs, and are likely to be exceeded if detected (except possibly N-nitrosodiphenylamine). However, only trace amounts of the carcinogens were detected and at only a few locations. PAHs are unfortunately relatively common in urban runoff, treated sewage, and some industrial wastes. They are mostly produced by the combustion of fossil fuels (including gasoline).

Phthalate esters, though relatively common in the samples, are not expected to exceed any of the applicable standards. However, many of the organic carcinogens detected, especially the PAHs, are expected to exceed the fish and water consumption standards and the fish consumption only standards at several locations in the watershed area studied. Like arsenic, another carcinogen, these data indicate the need to more fully monitor these important human health pollutants.

Water Quality Conditions at Mid-Watershed Locations in the Cahaba River

Data were examined for two mid-watershed locations in order to demonstrate the general nature of the existing receiving water problems in the Cahaba River. The stations were selected based on the availability of data and potential upstream pollutant sources. The concentrations of dissolved arsenic, cadmium, lead, mercury, and zinc in the Cahaba River at these locations frequently exceed the fresh water aquatic life criteria and the human health criteria for the consumption of fish. These data indicate that these problems are not localized.

The following table summarizes Cahaba River water quality, as monitored at the West Blocton and Centreville stations, as recorded in the EPA's STORET data file for the period of 1970 through 1989:

West Blocton Station:

Constituent and units (μg/L, unless noted)	Number of obs.	Mean	Standard deviation	Max.	Min.
Chlorides, mg/L	34	5.3	3.0	11	1
Specific conductance, μmhos/cm	58	217	77	380	58
Arsenic, dissolved	30	1.7	1.7	6	nd
Cadmium, dissolved	30	1.1	0.96	4	nd
Chromium, dissolved	30	1.2	2.1	8	nd
Iron, dissolved	45	66	72	420	10
Lead, dissolved	30	4.1	4.9	24	nd
Manganese, dissolved	45	29	33	220	nd
Mercury, dissolved	31	0.13	0.11	0.4	nd
Strontium, dissolved	32	96	45	190	20
Zinc, dissolved	32	15	16	60	nd

Centreville Station:

Constituent and units (μg/L, unless noted)	Number of obs.	*Mean*	Standard deviation	Max.	Min.
Chlorides, mg/L	61	3.3	1.4	7	1
Specific conductance, μmhos/cm	151	196	55	295	24
Arsenic, dissolved	40	1.5	1.5	7	nd

Cadmium, dissolved	48	2.4	3.8	25	nd
Chromium, dissolved	48	1.3	1.6	8	nd
Iron, dissolved	46	92	97	480	nd
Lead, dissolved	48	1.7	1.9	9	nd
Manganese, dissolved	54	27	27	160	nd
Mercury, dissolved	39	0.21	0.21	1.2	nd
Strontium, dissolved	48	61	32	140	nd
Zinc, dissolved	43	36	73	440	nd

note: “nd” is not detected, and a detection limit was not given.

The Centreville data indicated higher concentrations for cadmium, mercury and zinc, compared to West Blocton. West Blocton lead and strontium concentrations were periodically greater than those recorded at Centreville. The other pollutant concentrations were about the same at both the Centreville and West Blocton locations.

The water quality data contained in the above summaries were collected during the time period from 1970 through 1989. Plots of concentrations with time show that the spread of observed concentrations was consistent over the years and statistical tests showed that no significant trends in quality occurred. However, the data in the most recent years were only collected a few times a year, making trend analyses difficult. Obviously, reductions in discharges in many point source pollutant sources (especially from industrial and municipal wastewater treatment facilities) have occurred during this time. Unfortunately, many new sources, especially new municipal wastewater flows along with increased urban runoff and construction erosion runoff, has also occurred during this period of time.

It is apparent that some of the concentrations of toxic metals in the Cahaba River have historically exceeded the water quality criteria. All of the data for the heavy metals of potential concern were evaluated for the West Blocton and Centreville sampling locations on the Cahaba River. Many of the criteria for aquatic life are dependent on water hardness. Therefore, individual criterion were calculated for each data observation, depending on the water hardness also observed at each date. This allowed a determination of the frequency of criteria violations to be made for each pollutant. Figures 11 through 15 are probability plots showing the frequency of exceedences of applicable standards. The cadmium, lead, and zinc plots show several lines representing standards calculated using the 10th, 50th, and 90th percentile hardness values. The plots are truncated representing the limits of the observations based on the detection limits. In all of these cases, the criteria were exceeded for relatively high fractions of the samples obtained. The following list summarizes these criteria violations (the percentage of the data observations that exceeded the criterion, and the maximum ratio of the observed concentrations to the criterion value) for these two data collection sites, using actual simultaneous hardness values.

	Frequency and Exceedence of Violations:	
	West Blocton	**Centreville**
Arsenic		
human health (fish consumption) (EPA Criterion only)	100% (45X)	100% (40X)
Cadmium		
chronic aquatic life	55% (6.3X)	40% (10X)
acute aquatic life	5% (2.3X)	2% (4.9X)
Lead		
chronic aquatic life	54% (17X)	29% (20X)
Mercury		
human health (fish consumption)	50% (2.7X)	61% (20X)
chronic aquatic life	100% (33X)	100% (240X)
acute aquatic life		2% (1.2X)
Zinc		
chronic aquatic life		5% (3.3X)
acute aquatic life		5% (3X)

Figure 11. Probability plot of observed arsenic concentrations at Centreville compared to applicable criteria.

Figure 12. Probability plot of observed cadmium concentrations at Centreville compared to applicable criteria.

Figure 13. Probability plot of observed lead concentrations at Centreville compared to applicable criteria.

Figure 14. Probability plot of observed mercury concentrations at Centreville compared to applicable criteria.

Figure 15. Probability plot of observed zinc concentrations at Centreville compared to applicable criteria.

Serious violations of the human health criteria were observed for arsenic and mercury. Both the mean and maximum observed dissolved arsenic concentrations at Centreville (1.5 and 7 µg/L, respectively) greatly exceeded the EPA human health criterion for fish consumption (0.175 µg/L). This criterion was violated by every sample collected at West Blocton and Centreville.

The mean and maximum dissolved mercury concentrations (0.21 and 1.2 µg/L, respectively) greatly exceeded the Alabama state chronic aquatic life criterion of 0.012 µg/L. All samples collected at both West Blocton and Centreville exceeded this criterion. Both the average and maximum observed dissolved mercury concentrations also exceeded the EPA fish consumption human health criterion for mercury (0.146 µg/L). Half of the West Blocton samples exceeded this criterion, while more than 60 percent of the Centreville samples exceeded this criterion.

Figures 16 through 19 are plots of the ratios of the observed concentrations to the applicable water quality criterion, as a function of time. No statistically significant trends in the magnitudes or frequencies of violations has apparently occurred over the twenty years of record. It is also obvious that the chronic aquatic life criteria for lead and mercury were exceeded much more than once every three years, the suggested EPA allowable frequency standard.

Figure 16. Ratios of observed concentrations of arsenic to human health, fish consumption criterion.

Figure 17. Ratios of observed concentrations of lead to chronic aquatic life criterion.

Figure 18. Ratios of observed concentrations of mercury to human health, fish consumption criterion.

Figure 19. Ratios of observed concentrations of mercury to chronic aquatic life criterion.

Recent Water Quality Conditions Observed in the Upper Cahaba River

During the summer of 2000, a short 6-week survey of water quality conditions in the upper Cahaba River was undertaken by faculty and students from UAB and Miles College, including high school summer interns. The original intention of this project was to compare nutrient, selected pesticide and turbidity conditions at 11 sampling locations during wet and dry weather conditions. Because of the extended drought this summer, almost all of the samples were obtained during dry weather. However, during a few sampling periods, some of the monitoring stations were affected by rainfall runoff.

Samples of the Cahaba and the Little Cahaba Rivers and two tributaries were taken from the following eleven sites:

Little Cahaba River
Moody Highway 411 (below the Moody Wastewater Treatment Plant)
Leeds Wastewater Treatment Plant (above the treatment plant)
Leeds Ballpark (below the Leeds Wastewater Treatment Plant)
Lake Purdy Boat Launch
Cox Creek (a tributary)
Lee Branch (a tributary – sample taken above a nursery)
Cahaba Beach Road
Cahaba Pumping Station

Cahaba River
Highway 78
Grant Mills Road
Liberty Park
Cahaba Pumping Station (same as listed on the Little Cahaba) – water at the intake is a mix of both rivers

Methodology

This project was performed by two teams: one from Miles College and one from The University of Alabama at Birmingham. Grab samples were collected by each team twice per week (on the same day) and, when necessary, as soon as possible after a rainfall-runoff event for the six-week period from June 16 through July 27, 2000. 500 mL of samples were collected in amber glass jars with Teflon-lined lids. The samples were returned to the UAB laboratory and were analyzed for the parameters and using the methods listed in Table 8. Table 9 lists the potential health effects caused by the pollutants of interest. It also lists the maximum levels of each contaminant that is allowed in the finished drinking water.

Table 8. Laboratory Analyses

Analytes	Analysis Method
2,4-D	Elisa Method with Hach DR\2010 Spectrophotometer
Ammonia	EPA Method 350.2
Chlordane	Elisa Method with Hach DR\2010 Spectrophotometer
Chlorpyrifos	Elisa Method with Hach DR\2010 Spectrophotometer
Conductivity	Horiba Conductivity Meter
Nitrate	EPA Method 353.5
pH	EPA Method 150
Phosphate	EPA Method 365.2
Sulfate	Standard Methods Method 4500 – SO42 – E
Turbidity	EPA Method 180.1
Conductivity	EPA Method 120.6

Source: *Standard Methods for the Examination of Water and Wastewater, 19th Edition*. APHA, AWWA, and WEF. 1995.

Table 9. Potential Health Effects and Drinking Water MCLs for the Pollutants of Interest

Pollutant	**Potential Health Effects**	**Maximum Contaminant Level (MCL)**
Nitrate	Causes methemoglobinemia ('blue baby disease') in infants	10 mg/L
2,4-Dichlorophen-oxyacetic acid (2,4-D)	Pancreatic damage, Central Nervous System effects, Mutagenicity, and Teratogenicity.	70 μg/L (proposed*)
Chlordane	Upper respiratory tract irritation, liver damage, Peripheral nervous system effects, Embryo toxicity, and Carcinogenicity	2 μg/L (proposed*)
Ammonia	Eye, skin, and upper respiratory tract irritation, allergic sensitization, central nervous system effects	NA
Chlorpyrifos	Skin irritation and liver damage	NA
Sulfate	Laxative effect	250 mg/L (Secondary Drinking Water Standards)
Turbidity		1 – 5 NTU

*Source: *The Water Encyclopedia, 2nd Edition*. Edited by Frits van der Leeden, Fred L. Troise, and David Keith Todd. Lewis Publishers, Boca Raton, FL. 1990.

Results

Samples were collected from June 16, 2000 through July 27, 2000. Figure 20 shows the rainfall and the sampling dates. Generally for rainfall events, samples were collected either that day or on the morning after the event. No rain was recorded by the National Weather Service between June 29, 2000 and July 15, 2000. This provided the opportunity to investigate the chemical quality of the river during drought conditions.

Figure 20. Rainfall and Sampling Dates.

The following USGS flow monitoring data cover the main sampling period for four locations in the monitoring area. These data indicate the highly variable conditions found at the different locations throughout the watershed, and the poor representation of the Birmingham single rain gage data to indicate elevated flow conditions. This data also shows the very rapid rise that the Cahaba River experiences during moderate to heavy rainfall conditions.

Nutrients

Figures 21 through 23 show the results for nitrate, ammonia and phosphate, respectively. The nitrate concentrations for all sites along the Cahaba River were approximately 1 to 2 mg/L except for the Highway 78 site on June 29th (after a rain event) and on July 10th (during the middle of the drought). The concentration of nitrate below the Moody Wastewater Treatment Plant was slightly greater than 5 mg/L on June 20th. High concentrations of nitrate (about 5 mg/L) were also found around the Leeds Treatment Plant.

Figure 21. Nitrate concentrations by location along the Cahaba River and the Little Cahaba River and its tributaries.

Elevated ammonia concentrations were also seen on July 13th at several locations, although no rain was recorded at the Birmingham rain monitoring station and the recorded USGS streamflow for the Cahaba River remained low for that sampling period. The Little Cahaba River had greatly elevated ammonia levels on June 29th, corresponding to a moderate rain and elevated flows. The ammonia concentrations on the Cahaba River were elevated that day also, but not by as large a factor.

Figure 22. Ammonia concentrations in the Cahaba and Little Cahaba.

Elevated phosphate concentrations were seen in both rivers after rainfall events. This was especially evident for the Highway 78 site on June 29th and the Moody site on July 27th.

Figure 23. Phosphate concentrations in the Cahaba and Little Cahaba.

Pesticides

The results for 2,4-D, chlordane and chlorpyrifos (Dursban™) are shown in Figures 24 through 26. While detectable concentrations of 2,4-D are found in both the Cahaba and Little Cahaba Rivers, the concentrations are well below the MCL for drinking water. The pesticide concentrations were much larger during periods affected by rainfall, especially for 2,4-D.

Figure 24. Concentrations of 2,4-D in the Cahaba and Little Cahaba Rivers.

The concentrations of chlordane in both the Little Cahaba and Cahaba River exceeded the MCL on June 29th at all sites along the Cahaba River and at Cahaba Beach Road and the Pump Station Intake. On July 10th, all sites along the Little Cahaba River had elevated concentrations of chlordane that exceeded the MCL.

Figure 25. Chlordane concentrations in the Cahaba and Little Cahaba.

The concentrations of chlorpyrifos were also elevated in the Cahaba River both on June 29th and July 10th. They were also elevated along the Little Cahaba River on July 10th, with the exception of the Cahaba Beach Road site.

Figure 26. Chlorpyrifos concentrations in the Cahaba and Little Cahaba Rivers.

Turbidity

The results of the turbidity analyses are shown in Figure 27. The infrequent high levels of turbidity were likely associated with runoff from the land surrounding the river, especially nearby construction sites.

Figure 27. Turbidity in the Cahaba and Little Cahaba.

Allowable Wastewater Discharges

When determining allowable waste loadings that can be discharged to a receiving water, regulatory agencies consider the best levels of treatment economically achievable by the industrial category and the assimilative capacity of the receiving water. The discharge limits based on treatment levels defined by the industrial category are usually given as concentration limits in the discharge waters. The discharge limits established to protect the receiving water uses (the assimilative capacity) are based on critical low flows, uses of the receiving waters, existing background water pollutant concentrations, and expected future demands on the water. These assimilative capacity limits are usually given in allowable discharge mass limits (such as the maximum pounds per day of a pollutant that can be discharged). The following discussion shows that the receiving water assimilative capacity of the Cahaba River for toxic heavy metals is severely limited.

Critical Low Flows in the Cahaba River

The ADEM Water Division prepared a water quality toxicity policy report in September 1988 describing the calculation procedures for chemical specific limits that use the minimum 7-day average low flow value that occurs once in 10 years (7Q10). ADEM also states, in the Alabama *Toxic Pollutant Criteria Applicable to State Waters* (Code of Alabama 335-6-10-.07), that “For the purposes of establishing effluent limitations pursuant to Chapter 335-6-6 of the Department's regulations, the minimum 7-day low flow that occurs once in 10 years (7Q10) shall be the basis for applying the chronic life criteria, and the minimum 1-day low flow that occurs once in 10 years (1Q10) shall be the basis for applying the acute aquatic life criteria.” ADEM further states (Code of Alabama 335-6-10-.07), that “For the purposes of establishing effluent limitations pursuant to Chapter 335-6-6 of the Department's regulations, the minimum 7-day low flow that occurs once in 10 years (7Q10) shall be the basis for applying the human health criteria for pollutants classified as non-carcinogens, and the mean annual flow shall be the basis for applying the human health criteria for pollutants classified as carcinogens.”

As an example, average flows must therefore be used for arsenic discharge limitation calculations using the 0.175 ug/L receiving water criterion for the consumption of fish alone. The discharge limitation calculations for the chronic freshwater fish and wildlife criteria for arsenic, cadmium, chromium, lead, mercury and zinc should use the lowest 7 day average flows that occur once in ten years (7Q10). Calculations for discharge limitations for the acute freshwater fish and wildlife criteria for these metals should use the lowest 1 day average flows that occur once in ten years (1Q10). The following summarizes these flow values (in cubic feet per second, or CFS) for the Centreville location:

	Expected Flows (CFS)		
	Average	**7Q10**	**1Q10**
Cahaba River at Centreville	1607[1]	140[2]	<<140[3]

footnotes:

(1) 60 year average, presented in *Water Resources Data for Alabama* (USGS, 1987 water year).
(2) The 7Q10 is from the *Cahaba River Basin Water Quality Management Plan* (Alabama Water Improvement Commission, July 1974).
(3) The 1Q10 values are assumed to be substantially less than the 7Q10 values.

7Q10 values for other locations along the Cahaba River have also been tabulated in the *Cahaba River Basin, Water Quality Management Plan*:

Station number	Closest upstream tributary	Drainage area (square miles)	7Q10 (CFS)
18	Black Creek	115	7.0
16	Little Shades Creek	230	0
14	Buck Creek	unknown	25
6	Schulte Creek	1029	150
5	Dobine Creek	1379	220
1	Oakmulgee Creek	1768	280

All of the locations downstream of station number 18 are affected by the withdrawals of drinking water from Lake Purdy and the Cahaba River by the Birmingham Water Works Board (BWWB). These withdrawals averaged 82 CFS during the 1987 water year. Therefore, the 7Q10 values below this location are less than what would naturally occur. During critical low flow periods, the BWWB withdrawals reduce the Cahaba River flow to zero below their highway 280 diversion dam.

Unfortunately, low flow values for most locations along the Cahaba River, and especially its tributaries, are not readily available. The 7Q10 values for small tributary creeks can be estimated by multiplying their drainage areas by the unit area 7Q10 values contained in *The Map Abstract of Water Resources, Alabama* (Alabama Development Office, University of Alabama, and Geological Survey of Alabama, 1974). The 1Q10 values would be much less than the 7Q10 values.

Water Quality During Critical Low Flow Conditions

The specific conductance values, along with the concentrations of many of the pollutants, in the Cahaba River, are generally inversely related to the river flow. The river specific conductance values are greatest when the river flows are the lowest. Discharge limits must therefore consider these higher background values, which occur during the critical low flow periods, when determining discharge limits for the pollutants.

A plot of all of the flow and specific conductance values obtained at the Centreville site, from 1970 through 1989 (the complete data record contained in the EPA's STORET data file for this location as of 1990) shows that the critical low flow periods are strongly associated with the highest specific conductance values. Even though the mean specific conductance value at this location is about 200 μmhos/cm for this time period, the low flow periods had specific conductance values as much as 1.6 times as great (up to 315 μmhos/cm). The highest concentrations of other pollutants also occur during the critical low flow periods (generally between 100 and 1,000 CFS), as shown on Figures 28 through 30 for arsenic, chromium, and lead. Therefore, besides having very low flows, it is likely that many of the critical periods will also be associated with higher than average pollutant concentrations, resulting in reduced assimilative capacity in the receiving water.

Figure 28. Arsenic vs. flow at Centreville.

Figure 29. Chromium vs. flow at Centreville.

Figure 30. Lead vs. flow at Centreville.

Discharge limits calculated for critical low flow periods must consider the higher than average background Cahaba River pollutant concentrations during these critical periods. Dilution factors, if available to produce acceptable in-stream concentrations, would be the lowest during the low flow periods.

Allowable Discharge Limits

Critical flow and background pollutant conditions need to be used to calculate discharge limits for treated wastewaters. The following table summarizes the pollutants of concern, along with the most critical applicable water quality criterion, and the maximum dissolved pollutant concentrations observed in the Centreville vicinity:

Pollutant	**Critical Criteria Conc. (μg/L)** [1]	**Max. Background Conc. (μg/L)**
Chlorides	230 mg/L (aquatic life)	7 mg/L
Arsenic	0.175 (fish consumption)	7
Cadmium	0.99 (chronic aquatic life)	25
Chromium (Cr^{+6})	11 (chronic aquatic life)	8 (likely Cr^{+3} and not Cr^{+6})
Lead	2.5 (chronic aquatic life)	9
Mercury	0.012 (chronic aquatic life)	1.2
Zinc	91 (chronic aquatic life)	440

(1) Based on the median hardness concentrations observed for cadmium, lead, and zinc.

Therefore, the only pollutants shown on the above list that can be permitted to be discharged into the Cahaba River in this area at all are chlorides and chromium. All of the other pollutants already substantially exceed the most critical criterion applicable and cannot be discharged. The following table indicates the total allowable discharges for these critical pollutants, the maximum background discharges observed, and the available discharge balance that can be allocated to future dischargers near Centreville (all in pounds per day):

Pollutant	Flow Condition	CFS	Maximum Allowable Discharge	Maximum Background Discharge	Available Discharge Balance
Chlorides	7Q10	140	173,000	5,270	168,000
Arsenic	Qavg	1607	1.5	13(1)	-11
Cadmium	7Q10	140	0.75	19	- 18
Chromium(2)	7Q10	140	28.9	6.0	22.0
Lead	7Q10	140	1.9	7	- 5.2
Mercury	7Q10	140	0.009	0.9	- 0.9
Zinc	7Q10	140	70	330	- 260

(1) based on average arsenic concentration of 1.5 ug/L because arsenic is a carcinogen and the criterion is applicable for average flow conditions.

(2) assuming hexavalent chromium, which is unlikely.

Similar calculations could be made for other locations to determine allowable daily discharge limits.

Preliminary Assessment of Water Pollutant Sources in the Cahaba River Basin

Sources of Pollutants in the Upper Cahaba River

There are a number of pollutant sources in the Cahaba River above the Highway 280 crossing. The Alabama Department of Environmental Management administers the EPA's discharge permit (NPDES) program. These permits have conventionally been issued to point sources of pollutants, such as industrial facilities and sanitary wastewater treatment plants that discharge pollutants into waters of the state. The permits contain discharge limits, either as concentration limits or quantity limits, that are intended to allow pollutant discharges without causing violations of water quality standards. The following dischargers have discharge permits (in 1990) in the Cahaba River, or tributaries, above the Highway 280 crossing:

Trussville Industrial Park sewage treatment plant
Trussville municipal sewage treatment plant
Amerex Corp., Trussville
Gold Kist Poultry, Trussville
Riggins Tallow Co., Trussville
Southeast Bumper Distributors, Inc., Trussville
Hallmark and Son Coal Co., Henry Ellen Mines
Mann Steel Products, Inc., Henry Ellen Mine #2
Nugget Coal Company, Inc., Peacock Mine

These permits include two small Trussville area sewage treatment plants, three industrial facilities (including a manufacture of fire extinguishers; another doing metal work, including plating; and a poultry packaging facility), and three coal mines. The types of pollutants from these facilities are therefore very broad, including nutrients, solids, and heavy metals.

The watershed has about 185 square miles above the Highway 280 crossing, not considering the Little Cahaba River watershed. This area is mostly forested, but includes a growing amount of urban lands. Trussville, plus parts of Mountain Brook, Irondale and Roebuck Plaza, are the major urban areas in the watershed. A number of small unincorporated communities are also located in this upper watershed area. These urban lands make up an estimated ten square miles of the watershed area (including about one square mile of land undergoing development, or cleared for construction), leaving about 175 square miles of forested lands. There are no significant row crops, feed lot operations, or orchards in this area. Other nonpoint activities in the watershed include various mining operations, included in the above NPDES permit listing. Several abandoned coal mines and landfills are in the watershed that could be contributing significant water pollutants, but specific data are not available.

Table 10 summarize the known nonpoint discharges (in tons per year) for the estimated nonpoint and the NPDES permit discharges for the upper Cahaba River.

Table 10. Estimated Sources of Water Pollutants in the Upper Cahaba River (tons/yr)

	Forestry	Urban Const.	Mixed Urban	NPDES(1)	Total Expected
Suspended Solids	50,000	15,000	750	75	65,825
COD	200	25	250	n/a	475
Total Phosphorus	1.5	4	2	n/a	7.5
Total Nitrogen	65	25	10	8(2)	108
Arsenic	0	0	0.05	n/a(3)	0.05
Cadmium	0	0	0.06	n/a	0.06
Chromium	0	0	0.2	n/a	0.2
Copper	0	0	0.3	n/a	0.3
Lead	0	0	2	n/a	2
Zinc	0	0	2	n/a	2

(1) BOD_5 annual discharges total 50 tons/yr and Oil and Grease limits are 25 tons/year. There are no NPDES discharge quantity limits set for any other pollutants, besides suspended solids and total nitrogen, as shown on this table.

(2) Total Kjeldahl nitrogen (ammonia plus organic nitrogen forms) only. Ammonia limits are set at 3 tons/year also (but are part of the TKN value).

(3) The NPDES permits do not contain discharge limits (or restrictions) for these pollutants.

The permitted NPDES discharges are a very small fraction of the total expected discharges for all pollutants. Forestry is likely responsible for most of the suspended solids, COD, and nitrogen discharges, while the general urban runoff category is likely responsible for most of the toxicant discharges. The NPDES dischargers probably contain some of the other pollutants, but the permits do not contain other restrictions or information. Other sources, such as abandoned mines and landfills, are also possible important sources of some of the problem pollutants.

Evaluation of Sources Affecting Lower Reaches of the Cahaba River

This section is a review of the land use and Alabama NPDES (National Pollutant Discharge Elimination System, the Federally required permit program to control water pollutant sources that is administered by ADEM in Alabama) discharge permit information available for the Cahaba River basin. The purpose of this evaluation was to obtain an estimate of the relative water pollutant contributions from various known point and nonpoint sources.

Fifteen municipal wastewater treatment facilities, 28 mining facilities, and 33 industrial facilities had Alabama NPDES permits in the Cahaba River basin in 1990, the time when this analysis was conducted. The following list shows those located above Centreville, the location where most of the lower watershed water quality data was obtained. These are the majority of the NPDES permitted facilities in the watershed. Some of these permits have been issued but have not been used. A number of pollutants are included in the discharge permits, but are not restricted by mass discharge (pounds per day). Many of the pollutants have effluent concentration limits alone (such as in mg/L). Without mass discharge limits (or volume limits in conjunction with the concentration limits), the amounts of pollutants that could be discharged by the permitted facility is not restricted.

- **Municipal Discharge Permits above Centreville (ADEM Discharge Permits as of 1990)**
  Cahaba River, Hoover
  Leeds
  Trussville
  Hoover
  Trussville (922934)
  Alabaster
  Helena
  Pelham
  Pelham, Hunters Glen
  Riverview, Birmingham
  Moody
  Centreville- Brent
  Montevallo

- **Mining Discharge Permits above Centreville (ADEM Discharge Permits as of 1990)**
  Blue Circle Inc., Roberta Plant
  Cheney Lime and Cement Co., Landmark Plant

Dravo Basic Materials Co., Maylene Plant
Lehigh Portland Cement Co., Leeds Plant
Vulcan Materials Co., Calera Quarry
Southern Ready Mix Inc., Calera Rock Quarry
Vulcan Materials Co., Helena Quarry
Bickerstaff Clay Products Co., Plant #5
Vulcan Materials Co., Parkwood Quarry
Ray Cisco Construction Co., Shale Pit
Alabama Refactory Clay Co., Montevallo Pit
Bickerstaff Clay Products Co., Plant #6
BWS Technology Inc., Blocton #9 Reclamation Project
Allied Products Co., Grayhill-Nunnally Mine
Faulkner Energy Corp., Gurnee Mine
Hallmark and Son Coal Co.
Mann Steel Products Inc., Henry Ellen Mine #2
Allied Products Co., Woods tock Pits #1 and #2
B and G Mining Co., Yeshic Mine
Central AL Paving Inc., White Pit
New Circle Inc., Overton
Nugget Coal Co., Peacock Mine
U.S. Steel Mining Co., Gurnee Mine

• **Industrial Discharge Permits above Centreville**
(ADEM Discharge Permits as of 1990)
A. J. Gerrard and Co.
Alabama Great South RR Norris Yard
Amerex Corp.
Birmingham Steel Drum
Cahaba Pressure Treated Forest Products
Colonial Pipeline Co., Pelham Junction
Electrical Specialty Products Co.
Gold Kist Hatchery
Gold Kist Poultry
Hawkeye Oil and Gas, Inc.
Interstate Lead
M and B Metal Products Co.
Met rock Steel and Wire Co.
Olon Belcher Lumber Co.
Owens-Illinois, Inc.
Riggins Tallow Co.
Rock Wool Manufacturing Co.
Seaman Timber Co.
Southeast Bumper Distributors, Inc.
Southern Ready Mix, Inc., Plant #1
Southern Ready Mix, Inc., Plant #2
Sprviell Dairy Farm, Inc.
Southern Precision Corp.
Square D Co., Anderson Plant
United Chair
Vulcan Metal Products, Inc.

The expected nonpoint source discharges associated with forestry operations in the Cahaba River basin were estimated based on land use information from a number of sources. The unit area discharges and deliveries were mostly obtained from the *Alabama Cooperative Study of the Alabama River*, by the USDA in 1977. The delivery values estimate the fraction of the source area sheetflows that actually reach the receiving water. The amount of the pollutants that actually travel down the river to downstream areas is a function of many in-stream processes. The values from these sources for suspended solids was about 1.7 tons per acre per year lost, with about 25 percent being delivered to the receiving water. The COD loss was about 15 pounds per acre per year, the total phosphorus loss was about 0.12 pounds per acre per year, and the total nitrogen loss was about 4.3 pounds per acre per year, all with an estimated 25 percent delivery to the receiving water.

Similar calculations were made for agricultural sources. The unit area pollutant rates were also obtained from the *Alabama Cooperative Study* and were based on the mixture of different crops expected in the river basin. The values from these sources for suspended solids was about 5 tons per acre per year lost, with about 25 percent being delivered to the receiving water. The COD loss was about 140 pounds per acre per year, the total phosphorus loss was about 0.46 pounds per acre per year, and the total nitrogen loss was about 9.9 pounds per acre per year, all with an estimated 25 percent delivery to the receiving water.

Analyses were also made of urban construction site discharges. The areas under construction were assumed to be about six percent of the total urban area, based on preliminary Birmingham area surveys (specifically the Rocky Ridge Corridor demonstration project). The discharges were obtained from many local Birmingham area samples and averaged about 150 tons per acre per year. The deliveries to the streams were estimated to be about 25 percent. The COD loss was about 500 pounds per acre per year, the total phosphorus loss was about 80 pounds per acre per year, and the total nitrogen loss was about 500 pounds per acre per year, all with estimated 25 percent deliveries to the receiving water.

Mixed urban area pollutant contributions in the watershed were also estimated. The mixture of land uses was obtained from local Birmingham data from detailed local measurements obtained in the Rocky Ridge Corridor demonstration project conducted by the Jefferson Co. SCS office. The unit area pollutant discharges were obtained from NURP information (EPA 1983) and local EPA sponsored research (Pitt, *et al.* 1995). The values from these sources for suspended solids was about 250 pounds per acre per year lost, with 100 percent being delivered to the receiving water. The COD loss was about 80 pounds per acre per year, the total phosphorus loss was about 0.5 pounds per acre per year, and the total nitrogen loss was about 3.5 pounds per acre per year. The deliveries of these pollutants is 100 percent because the yields were measured at the outfalls to the receiving waters. The following additional urban area pollutant mass discharges were also estimated (all pounds per acre per year): lead: 0.5; zinc: 0.7; chromium: 0.06; copper: 0.1; cadmium: 0.02; and arsenic: 0.015.

Suspended solids from the nonpoint sources will mostly accumulate in the river, while suspended solids from treated wastewaters are more likely to travel greater distances downstream due to the settling characteristics of the particles. Most other pollutants are likely to travel more efficiently than the suspended solids. The following summary shows the estimated nonpoint discharges from these sources for the complete watershed:

Source	**Area (square miles)**	**Suspended Solids (tons/yr)**	**COD (tons/yr)**	**Total P (tons/yr)**	**Total N (tons/yr)**
Forestry	1350	370,000	1,600	13	470
Agriculture	130	100,000	1,400	5	100
Construction	13	300,000	500	80	500
Mixed urban	200	16,000	5,300	40	230
NPDES permits	-	320	400 (est)	NA	60

The following list summarizes the estimated pollutant contributions from the most significant sources for the whole Cahaba River watershed:

TSS: construction site erosion: 40 percent.
forestry: 50 percent.
COD: mixed urban runoff: 60 percent.
forestry: 20 percent.
Total P: construction site erosion: 60 percent.
mixed urban runoff: 30 percent.
Total N: forestry: 35 percent.
construction site erosion: 40 percent.
mixed urban runoff: 20 percent.
Lead: mixed urban runoff: about 100 percent.
Zinc: mixed urban runoff: about 100 percent.

The mass pollutant contributions from the NPDES permitted discharges are quite low:

TSS: <1 percent
COD: 4 percent.
Total P: N/A. Obviously, the municipal wastewater facilities are significant phosphorus dischargers, but none of them have P mass discharge limits set in their permits.
Total N: 4 percent.
Lead: <<1 percent.
Zinc: <<1 percent.

Estimated pollutant concentrations were calculated using these discharge values and an average river flow value of 3.8 X 10^{11} gallons per year at Centreville. These calculated concentrations were compared to the observed concentration values in the river as a rough check for the above mass balances. The following list summarizes these calculated concentrations and comparisons:

- TSS: 330 mg/L. This value is too high. It is expected that about 90 percent of the nonpoint suspended solids discharges would settle out in the river as sediment accumulations. The long-term average concentrations at Centreville are about 25 mg/L.
- COD: 4.5 mg/L. This value is close to the expected COD concentration at Centreville.
- Total P: 0.08 mg/L. This value is close to the expected P concentration at Centreville.
- Total N: 0.6 mg/L. This value is close to the expected N concentration at Centreville.
- Lead. 20 μg/L. This value is close to the expected total lead concentration at Centreville. The dissolved lead concentration at Centreville is about 2 μg/L. About ten percent of the total lead concentration is expected to be dissolved. The urban contribution of lead at Centreville is about 2.3 tons per year, while the total river flow accounts for about 2.7 tons per year.
- Zinc: 30 μg/L. This value is close to the expected total zinc concentration at Centreville. The dissolved zinc concentration at Centreville is about 35 μg/L. Usually almost all of the total zinc in urban runoff is in the dissolved form. The urban contribution of zinc at Centreville is about 39 tons per year, while the total river flow accounts for about 57 tons per year.
- Chromium: 2.5 μg/L. This value is about twice the expected total chromium concentration at Centreville. The dissolved chromium concentration at Centreville is about 1.3 μg/L. The urban contribution of chromium at Centreville is about 2.1 tons per year, while the total river flow accounts for about 2.0 tons per year.
- Cadmium: 0.8 μg/L. This value is about one-third the expected total cadmium concentration at Centreville. The dissolved cadmium concentration at Centreville is about 2.4 μg/L. The urban contribution of cadmium at Centreville is about 0.56 tons per year, while the total river flow accounts for about 3.8 tons per year. Other unaccounted sources, including natural background conditions, may be responsible for the remainder.
- Arsenic: 0.6 μg/L. This value is about one-half the expected total arsenic concentration at Centreville. The dissolved arsenic concentration at Centreville is about 1.5 μg/L. The urban contribution of arsenic at Centreville is about 0.49 tons per year, while the total river flow accounts for about 2.4 tons per year. Other unaccounted sources, including natural background conditions, may be responsible for the remainder.

These calculated concentrations generally agree with the observed concentration values from Centreville. Therefore the mass balances given above are also expected to be reasonable accurate. However, unaccounted sources of some of the heavy metals, especially cadmium (two-thirds) and arsenic (one-half), may be significant.

Cahaba River Conclusions

As indicated in the above discussion, high concentrations of many pollutants investigated were generally found in the highest reaches of the Cahaba River for which data was available. These were mostly located near likely localized sources (especially an abandoned landfill, old mines, and improper disposal at an industrial facility) and in stream reaches having relatively low flows. However, adverse concentrations for many constituents (especially heavy metals) were found at locations much further downstream from these localized sources. Cahaba River water quality at the Birmingham Water Works pumping station and even further downstream at Centreville indicated several problems described below. At these locations, urban stormwater was the largest source of pollutant discharges (compared to forestry operations, agriculture operations, and permitted NPDES discharges).

Pollutant Sources

The following graphs summarize some of the historical and more recent water quality observations for nutrients in the Cahaba River, including source area sheetflow concentrations (Pitt, *et al.* 1995) and land use yields (Lalor, *et al.* 1998). There is a wide range of concentrations of nutrients in the river, based on historical and recent observations. Much of the variation is related to wet weather flows, which can have nutrient concentrations many times greater than dry weather flows. In addition, some source areas contribute much greater amounts of nutrients than other areas. Roof runoff is quite low, while paved paring and storage areas can have significantly elevated concentrations of phosphorus. However, roof runoff and landscaped areas can have significantly elevated concentrations of ammonia, especially compared to many paved areas. In contrast to other areas, Birmingham area golf courses are seen to have much elevated concentrations of nutrients. In many areas, golf course runoff management and nutrient applications result in significantly reduced nutrient discharges compared to other sources. Another interesting local observations was the periodic very elevated concentrations of nutrients from some commercial areas (noted during periods when outside seasonal plant sales were being conducted).

Upper-Reach Cahaba River Conditions

The most serious water pollution problems in the upper reaches of the Cahaba River, in possible order of importance, are as follows:

- Toxicants exceeding the human health criteria (especially the carcinogens arsenic and the organic PAHs, plus mercury which always exceeded the EPA criteria, plus likely frequent violations of cadmium).
- Nuisance conditions, especially eutrophication (nutrient enrichment) caused by high phosphorus concentrations. Increased nitrogen discharges will dramatically worsen eutrophication conditions. Other nuisance conditions frequently occur caused by high iron content in drinking water and frequently turbid water.
- Aquatic life problems caused by toxicants (especially cadmium, lead, and possibly zinc), high temperatures during spawning periods, sedimentation because of localized high erosion, and infrequent low dissolved oxygen concentrations.
- Swimming problems caused by periodic exposures to pathogenic microörganisms (as indicated by high populations of fecal coliform bacteria).

The above problems are quite widespread and do not appear to be associated with specific locations, or times, although many of the toxicant problems are worse further upstream. Other problems identified (especially for chlorides and ammonia) are likely infrequent and were probably associated with intermittent (illegal?) industrial discharges. Some problems are also likely associated with more continuous discharges from improperly operated facilities (such as a landfill slightly above the BWWB pump station, upstream mines, and poorly operated wastewater treatment facilities).

Mid-Reach Cahaba River Conditions

The current high concentrations of several heavy metals in the Cahaba River indicate the need for serious further investigations. Numerous and large violations of the EPA arsenic and mercury human health criteria (fish consumption) were observed for previously collected Cahaba River samples. These criteria were violated by every sample collected at West Blocton and Centreville. These criteria exceedences signal the need for comprehensive fish (and other organism) tissue analysis to quantify the threat to human health.

All samples collected at both West Blocton and Centreville also exceeded the Alabama state chronic aquatic life criterion for mercury. Besides arsenic and mercury, other heavy metal pollutants of concern include cadmium and lead, because of significant aquatic life chronic criteria exceedences. The preliminary analysis of sources of pollutants into the river indicated that stormwater is the likely major source of most of the heavy metals. It is also expected that old mining operations or natural erosion through coal seams along the river may also be responsible for some of the metals found.

Overall

The major water quality problems in the Cahaba River are likely associated with heavy metals, nutrients, sediment, and oxygen demanding materials (in general order of exceedences of criteria). These problems are expected to change in future years as changes in the water pollution sources occur. The most dramatic change will be associated with urban development in the watershed. This can generally lead to increased discharges of treated sanitary wastewater, erosion materials, and urban runoff pollutants. It is hoped that the current problem pollutant sources (such as the improper sources mentioned above) will be corrected and future development will occur with care to minimize additional discharges. However, it must be recognized that additional pollutant discharges are inevitable, even with the best controls in place. The most significant discharges related to the current problems will likely be sediment during construction; plus heavy metal and organic toxicants, pathogenic microörganisms, and nitrogen from urban runoff. Urbanization will also cause some heating of the river. With proper controls, the amount of urbanization possible before conditions become unbearable (economically untreatable as a drinking water source or catastrophic to fish and other aquatic life) will be much greater than if no controls were used.

Most of the pollutant discharges into the Cahaba River are from nonpoint sources of pollutants. Municipal wastewater treatment systems, mining operations, and industrial discharges controlled by the Alabama NPDES system likely account for only small portions of the total waste discharges into the Cahaba River. These sources are expected to contribute less than ten percent of the total COD and nitrogen discharges and less than one percent of the total suspended solids and heavy metal discharges into the river. Certainly, if these sources were uncontrolled, their contributions would be much greater. Forestry operations may contribute about 50 percent of the suspended solids, 20 percent of the COD, and 35 percent of the nitrogen discharges. Construction site erosion runoff may contribute about 40 percent of the suspended solids, 60 percent of the phosphorus (excluding the unknown contributions from municipal wastewater operations), and 40 percent of the nitrogen discharges into the river. Urban runoff may contribute about 60 percent of the COD, 20 percent of the nitrogen, and practically all of many of the heavy metals being discharged into the river. It is imperative that detailed investigations, using appropriate TMDL procedures, consider urban stormwater as a likely source of pollutants to the Cahaba River.

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