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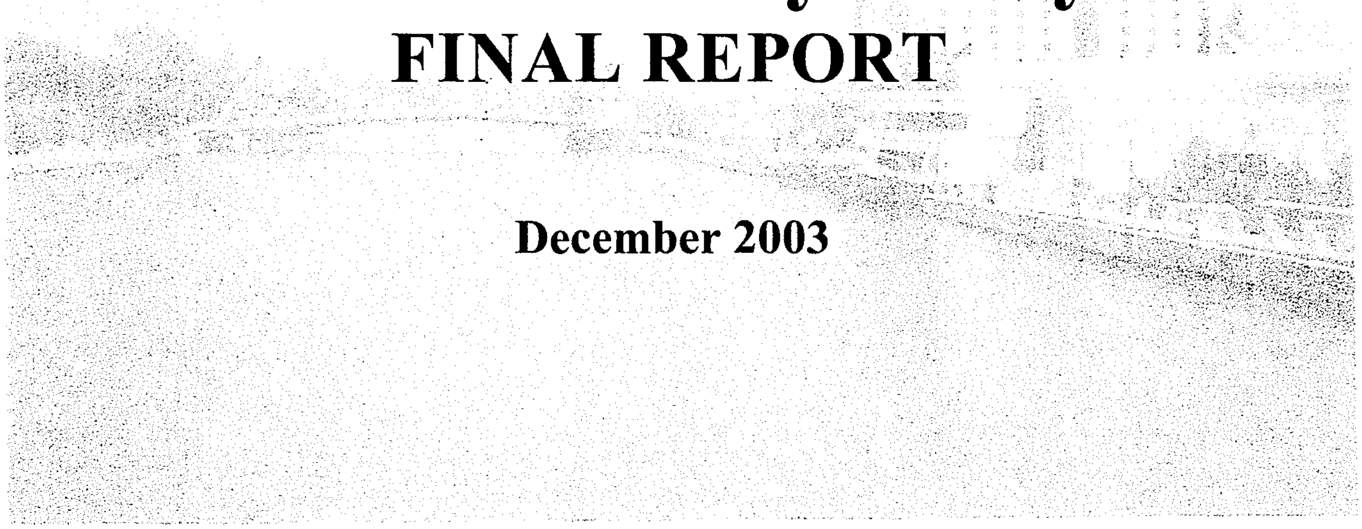
Hey and Associates, Inc.

Lower Des Plaines River
Use Attainability Analysis
FINAL REPORT



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and
Hey and Associates, Inc.

Lower Des Plaines River Use Attainability Analysis

FINAL REPORT

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CHAPTER 1

INTRODUCTION

The Need for the Use Attainability Analysis

This document presents the Use Attainability Analysis (UAA) for the Lower Des Plaines River in Illinois that has been classified by the state as a *Secondary Contact Indigenous Aquatic Life* use water body. The federal water quality standards regulation requires that states perform a Use Attainability Analysis (UAA) for water bodies where designated uses are lower than the statutory fish and aquatic life protection and propagation and primary contact recreation uses required by Section 101(a) of the Clean Water Act (CWA). In Illinois, the statutory use complying with the CWA goals is the *General Use.* The current other uses of the water body such as navigation, wastewater and storm runoff disposal may conflict with the higher statutory designated uses (aquatic life protection and propagation and contact recreation) represented by the General Use. The task of the UAA is to develop conditions for uses that would meet or approach aquatic life protection, propagation and primary recreation uses required by the Clean Water Act. Implementation of such standards is tested against the six reasons of the UAA regulations (Box 1.1), including avoidance of wide spread adverse socio-economic impacts that allow a downgrade of the use and/or of the standards or justify the standards that do not comply with the Illinois general use.

Watershed planning and management for control of all sources of pollution have been included in the Clean Water Act (Sections 208, 303, and 305) and subsequent regulations (40 CFR 130). In this context, the objective of watershed management is achieving water quality goals as expressed by the water quality standards and addressing pollution from all sources. There are two tools provided by the Clean Water Act and subsequent regulations that will initiate the process of watershed management. One is the Use Attainability Analysis (UAA) and the other is the Total Maximum Daily Load (TMDL).

The UAA requirement stems from Section 101(a) of the Clean Water Act that states: *it is the national goal that <u>wherever attainable</u> .. water quality provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water be achieved...* In this document we will refer to the uses in agreement with Section 101(a) as *statutory uses.* The General Use in Illinois is a statutory use. Consequently, the UAA study investigates whether the standards defining the designated use conforming with Section 101(a) of the CWA are attainable in the analyzed water body. If the statutory CWA use is not attainable, the UAA will define the most optimal attainable use for the water body.

On the other hand, the TMDL process is used for implementing state water quality standards, i.e., it is a planning process that will lead to the goal of meeting the water quality standards in *water quality limited receiving water bodies* and, *de facto,* it presumes that the statutory use is attainable. Both the TMDL and UAA may be prepared for individual waterbodies or their segments; however, the UAA should precede the TMDL. TMDL and UAA are performed for **water quality limited**

segments that have been specifically defined by the EPA as *those segments that do not or are not expected to meet applicable water quality standards even after the application of technology - effluent limitations required by Sections 301(b) and 306 of the Clean Water Act.*

There are three categories of classification of water quality limited water bodies based on the source of pollution: (1) water bodies impacted solely by point sources for which the mandatory point source controls will not result in attainment of water quality goals; (2) water bodies impacted by both point and nonpoint sources for which the attainment of water quality goals will not be achieved by application of mandatory point source controls and reasonable and economically efficient nonpoint source controls; and (3) bodies impacted by nonpoint sources only.

In 1983, after revising the Water Quality Standards Regulations (40 CFR 131), the Use Attainability Analysis (UAA) was made the standard procedure through which states were to gather and analyze data and document decision processes used to resolve questions about site-specific attainability of designated use classes. While the USEPA does not demand that its published UAA guidelines (USEPA, 1983a, 1984a,b, 1991, 1994) are followed, any process that a state develops to address attainability issues must be sufficient to meet the intent of the UAA guidelines. The rationale of the Use Attainability Analysis is included in the EPA's *Water Quality Standards Handbook* (USEPA, 1983b, 1994). The process which defines water quality standards (WQS) for any (navigable) water body must consider whether the designated uses are appropriate for the water body. The EPA Handbooks specify that attainability or non-attainability of designated uses and their relevant standards are judged based on physical conditions, natural or irretrievable chemical conditions, and widespread and substantial socio-economic impact (Box 1.1).

In order to carry out the socio-economic impact analysis outlined in Item 6 of Box 1.1, the load capacity of the water body may need to be determined and a waste load allocation performed (Novotny, et al., 1997).

The UAA generally answers the following questions about the condition of the water body:

a) What is the existing use to be protected?
b) What is the extent to which pollution (as opposed to physical factors) contributes to the impairment of a use?
c) What is the level of point source control required to restore or enhance the use?
d) What is the level of nonpoint source control required to restore or enhance the use?
e) What are the needed water body restoration (waste assimilative capacity enhancement) measures that would alter adverse physical conditions of the receiving water body that are impacting the aquatic habitat as well as meeting water quality standards?
f) What is the optimal water use of the water body as defined in the ecoregional context of attainable water quality?
g) What is the optimal use of the water body that would not impose widespread adverse socio-economic impacts on the population involved and society as a whole?

With exception of the Item g, this report will address the above issues.

Box 1.1	**Six reasons for a change of the designated use and/or water quality standards of a water body (40 CFR 131)**
(1)	Naturally occurring pollutant concentrations prevent attainment of the use; or
(2)	Natural, ephemeral, intermittent or low flow or water levels prevent the attainment of the use unless these conditions may be compensated for by the discharge of a sufficient volume of effluent discharge without violating State conservation requirements to enable uses to be met; or
(3)	Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place; or
(4)	Dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use; or
(5)	Physical conditions related to the natural features of the water body, such as the lack of proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or
(6)	Controls more stringent that those required by Sections 301(b)(1)(A) and (B) and 306 of the Act would result in substantial and wide-spread adverse social and economic impact.

The Use Attainability Analysis can result in the following possible outcomes:

(1) The designated use and corresponding standards are confirmed as attainable;
(2) The designated use is confirmed as attainable; however, standards are modified to reflect ecoregional and/or site-specific attributes;
(3) The designated use is modified or sub classified with corresponding modification of standards; or
(4) The designated use is upgraded based on existing or potential uses. The case of upgrading existing uses may involve water bodies which had previous water use assignments lower than those specified by the CWA or water bodies which subsequent to the use assignment were designated as Outstanding National Resources Waters.

While most of the potential UAA's may have been developed throughout the nation or needed for a reason of downgrading the use or adjusting the standards, the Illinois Environmental Protection Agency (IEPA), in the case of the Lower Des Plaines River, is looking for a way to upgrade the present lesser use of the river defined as "secondary contact recreation and indigenous aquatic life." This classification established an objective of protecting the *existing aquatic organisms and allow limited non contact recreational opportunities and avoid nuisance and aesthetically impaired conditions.* The agency wishes to achieve the highest attainable water use consistent as closely as possible with the goals of the Clean Water Act expressed in Section 101(a).

Urbanization combined with the effects of artificial channelization, such as in the Lower Des Plaines River, represents a challenge in the UAA. The Lower Des Plaines River has been modified by three dams and locks (Lockport Lock & Dam, Brandon Road Lock & Dam, and Dresden Island Lock & Dam) and receives flow from the Chicago Sanitary and Ship Canal that, during low flows, carries mostly treated sewage from the Chicago area. Other urban wastewater and urban runoff contributions are brought by the upstream Des Plaines River and from the city of Joliet, IL. Thus, the stream can be characterized as *effluent dominated.* The water quality regulations do not exclude the effluent dominated streams from compliance with the water quality standards unless Reasons 3 and/or 6 of the UAA regulations (Box 1.1) provide relief.

Objectives of the Study

The Illinois EPA wishes to elevate the present lesser use of the Lower Des Plaines River from *Secondary Contact Recreation and Indigenous Aquatic Life* to a higher use for balanced aquatic life, contact recreation and, also considering water supply, if it is an existing or potential use. The impetus for this UAA is Section 131.10(j) of the Water Quality Standards Regulations. Figure 1.1 shows the map of the investigated river and the UAA reaches.

The UAA is a legitimate means to strive for a higher use when the designated use is a lesser use than that specified by Section 101(a)(2) of the Clean Water Act. If actions needed to upgrade the river quality and habitat do not cause "a widespread and substantial adverse socio-economic impact," the higher use is considered attainable unless one of the remaining five reasons prevents the attainment of the use. Unlike TMDLs that focus only on waste load and load allocations, the UAA can venture further and suggest water body and riparian zone restoration in addition to further reduction of waste water discharges and BMPs for nonpoint pollution.

The objectives of the study were specified by the IEPA as:

1. Evaluate all available data to determine the physical, chemical and biological conditions of the waterway.

2. Determine potential to achieve and maintain higher value uses such as a diverse and balanced self supporting aquatic community and primary contact recreation.

3. Identify and characterize the relative significance of major stressors on the system including potential use impairment identified in the agency's April 1, 1998 Clean Water Act Section 303(d) List.

4. Assess available water quality and habitat management activities to eliminate or reduce system stressors.

5. Develop recommended use designations and affiliated water quality standards to achieve the highest attainable uses consistent with the Clean Water Act goals and Chapter 2 of the USEPA (1994) Handbook.

Description of the Lower Des Plaines River

Des Plaines River Watershed

The Des Plaines River originates in Wisconsin. In Illinois, the Des Plaines River Watershed covers a total of 854,669 acres in Lake, Cook, DuPage, and Will counties. The majority of the watershed is part of the greater Chicago metropolitan area and has been extensively developed for urban and industrial use. The remaining rural and agricultural lands are primarily in Lake and Will counties. Major streams which comprise the Des Plaines River Watershed include the Des Plaines River, the DuPage River, Cal Sag Cannel, Chicago Sanitary and Ship Canal, Salt Creek, Mill Creek, Indian Creek, Willow Creek, Lily Cache Creek, Grant Creek, Hickory Creek, and Spring Creek. A total of 685 stream miles was assessed within the watershed by the Section 305(b) study by the IEPA. The overall resource quality shown on Figure 1.1. assessed in the 1998 Illinois Section 305(b) report was "good" on 165 stream miles (24%), "fair" on 481 stream miles (70%), and "poor" on 39 stream miles (6%). The potential causes of water quality problems identified in the Illinois Section 305(b) and 303(d) reports are nutrients, pathogens, siltation, and habitat alterations attributed to municipal point source pollution, urban runoff, contaminated sediments and/or phosphorus attached to sediment particles, and hydrologic/habitat modifications, including flow alteration.

The Des Plaines River

The Des Plaines River originates just south of Union Grove, Wisconsin, and enters Illinois near Russell, Ill. From Russell, the Des Plaines flows in a southerly direction through Lake and Cook counties. Near Lyons, Ill., the Des Plaines turns to the southwest paralleling the Chicago Sanitary and Ship Canal (CSSC) in DuPage and Will counties until the confluence with the CSSC near Joliet, Ill. The Des Plaines continues southwest to the confluence of the Kankakee and the beginning of the Illinois River. The watershed area of the Des Plaines River excluding the CSSC is 13,371 mi^2 and the CSSC drainage is 740 mi^2. The total main stem length of the river in Illinois from the State border to the confluence with the Kankakee River is 110.7 miles. The long-term average discharge of the Des Plaines River at Riverside, IL is about 350 cfs. This can be compared with the capacity of the Stickney, IL waste water treatment plant operated by the Metropolitan Water Reclamation District of Greater Chicago (MWRD) of 1,200 mgd, which is equivalent to 1,033 cfs. Since other treatment plants of the Chicago metropolitan area also discharge into the CSSC, clearly, the lower segment of the Des Plaines River is effluent dominated under low and medium flow conditions.

In the 2002 305(b) report, 33.4 miles of the main stem of the Des Plaines River were rated as "fully supporting the aquatic life use ("good") and 77.3 miles as partially supporting ("fair"- green designation on Figure 1.1). In 1998 305(b), the section between the confluence of the river with CSSC at RM 290.1 and the Brandon Road Dam at RM 286 was ranked as "poor" (not supporting). In the 2002 report, degraded water quality was attributed to nutrients and siltation from municipal and industrial point source pollution, urban runoff, contaminated sediments, priority organics, metals, ammonia, TDS/conductivity, suspended solids, flow alteration, and habitat alteration. Most of Northeast Illinois, where the river is located, is an urbanized area with municipal point source pollution, hydrologic/habitat modifications, and urban runoff as major sources of pollution.

The Study Reach

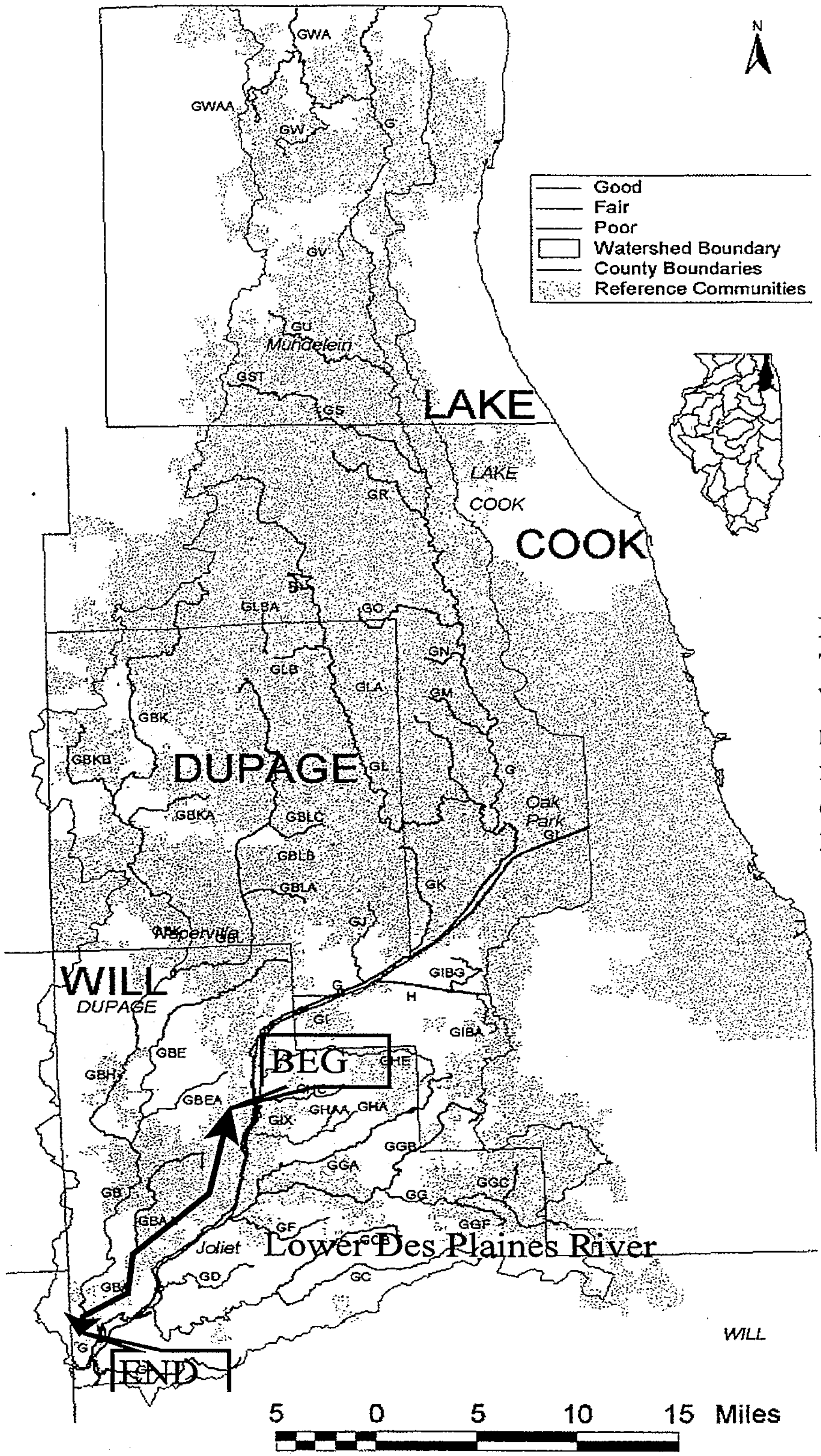

Figure 1.1
The Lower Des Plaines River Watershed and the UAA study reach. The figure is taken from the Illinois 305(b) water quality report (latest edition 2002)

The Use Attainability Analysis of the Lower Des Plaines River extends from the confluence of the river with the Chicago Ship and Sanitary Canal (CSSC) at the E.J.& E railroad bridge (River Mile 290.1 near Lockport) downstream to the Interstate 55 Highway Bridge at the River Mile 277.9 (Figure 1.1). Almost the entire reach is impounded and has two morphologically different segments, the Brandon Road Pool above the Brandon Road Lock and Dam (River Mile 286) and the portion of the Dresden Pool above the I-55 Bridge. The US Army Corps of Engineers operates the locks and dams to provide conditions for navigation (primarily barge traffic). The Lower Des Plaines River is on the Illinois EPA's Section 303(d) list of impaired waters.

The Brandon Road Pool is four miles in length, approximately 300 ft wide, with the depth varying between 12 - 15 feet. It is essentially a man-made channel that is bordered by side masonry, concrete or sheet pile embankments (Figure 1.2). The average velocity in the pool is 0.75 fps. The Chicago Sanitary and Ship Canal (CSSC) is the main tributary of the Lower Des Plaines River segment under consideration. The canal contributes approximately 80 % of flow to the river downstream from the confluence with the Des Plaines River. The water quality status of the Des Plaines River, upstream from the confluence with the Chicago Sanitary and Ship Canal, has been classified as fair. It receives urban runoff from many suburban communities. Runoff from the largest commercial diffuse source, the O'Hare International airport, is collected and conveyed to the Metropolitan Water Reclamation District for treatment.

The Dresden Island Pool is 14 miles long, approximately 800 feet wide, with the depth varying between 2 - 15 feet. The average stream velocity is 0.65 fps. The 8.1 miles reach of the impoundment that is a part of the UAA study is more natural than the Brandon Road Dam pool, meanders, and has a fair amount of natural shoreline and side channels (Figure 1.3). In the Dresden Island Pool, the US Army Corps of Engineers maintains a 9 foot deep navigational channel.

The Lower Des Plaines River is a part of the Upper Illinois Waterway. The Illinois Waterway is one of the busiest inland commercial navigation systems in the nation, providing a link between the Great Lakes/St. Lawrence Seaway navigation system and the Mississippi River navigation system that connects to the Gulf Intercoastal Waterway. The Illinois waterway includes the following segments:

- The Illinois River from its mouth at Grafton, IL to the confluence of the Kankakee and Des Plaines Rivers (273 miles)
- The Des Plaines River to Lockport Lock (18.1 miles)
- The Chicago Sanitary and Ship Canal which provides a connection to the deep draft system at Lake Calumet and Calumet Harbor, via the Little Calumet and Calumet Rivers (23.8 miles).

The entire waterway is completely channelized to a minimum depth of 9 ft and is used almost for commercial transport of bulk commodities such as grain, coal, petroleum products, chemical and raw materials.

Water Quality

Historically, the Lower Des Plaines River has received flows from the man-made Chicago Sanitary and Ship Canal which receives effluents from several Metropolitan Water Reclamation District of Greater Chicago wastewater reclamation plants and overflows from the combined sewers. Consequently, historically, the environmental potential of the river was deemed to be very limited to a point of hopelessness. The pollution population equivalent of effluent discharge carried by the canal to the Des Plaines River is about 9.5 million. The TARP project today has significantly reduced the number (frequency) of CSOs overflows per year. With the full implementation of the reservoir portion of TARP, the frequency of overflows will be further reduced. Combined sewer overflows reaching the river via the Chicago Sanitary and Ship Canal are a source of a mixture of untreated sewage and urban runoff from Chicago and Cook County.

Table 1.1 includes a list of large and medium size (more than 1 cfs) public wastewater treatment plants located on the Des Plaines River and the Chicago Waterways upstream of the I-55 bridge. It can be seen that the effluent discharges constitute the major part of the flow in the Lower Des Plaines River. The total effluent flow from the WWTPs is about 1900 cfs (1230 mgd) (Table 1.1). This effluent flow constitutes more than 90% of low flow in the Lower Des Plaines River and during winter, almost the entire low flow is made of effluent discharges. Consequently, the Lower Des Plaines is characterized as *an effluent dominated stream.*

Several large power plants use water from the CSSC and the Lower Des Plaines River for cooling. The thermal power plants operated by Midwest Generation are listed in Table 1.2 along with the power capacities and parameters. Two sites, Will County and Joliet #9 and #29 use most of the flow in the CSSC and the Lower Des Plaines River for cooling. During the summer of 1999, 24 supplemental cooling towers were installed at the Joliet Station #29 that are used on an as-needed basis to keep the temperature of the river at the I-55 bridge at or below the adjusted standard requested by Commonwealth Edison and approved by the State of Illinois Pollution Control Board.

Table 1.2 presents the heat release parameters of the power plants that may affect the temperature of the Lower Des Plaines River. By comparing the condenser cooling water flow and the river (canal) flow it becomes immediately apparent that two power production systems--Will County and Joliet power plants-- may use all of the flow of the Chicago Sanitary and Ship Canal (Will County) or the Lower Des Plaines River (Joliet) during low flow conditions.

The Illinois EPA 1998 303(d) list has identified the following parameters of concern for the sections between the confluence with the CSSC and the Kankakee River:

priority organics	*ammonia*
nutrients	*pathogens*
metals	*siltation*
habitat alterations	*flow alteration*
low dissolved oxygen/organic enrichment	

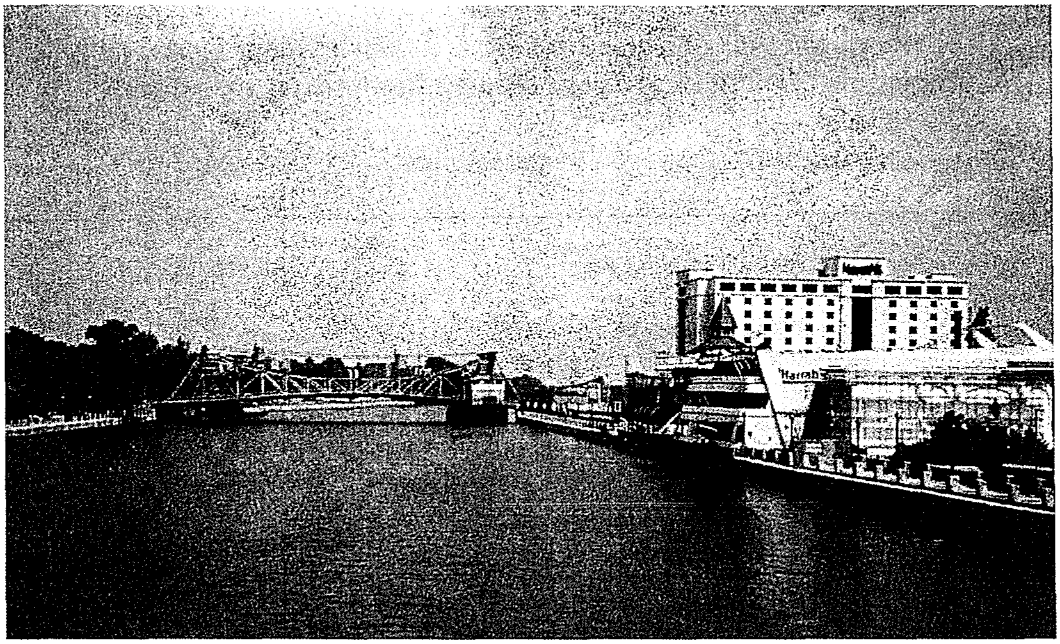

Figure 1.2 Brandon Pool in downtown Joliet

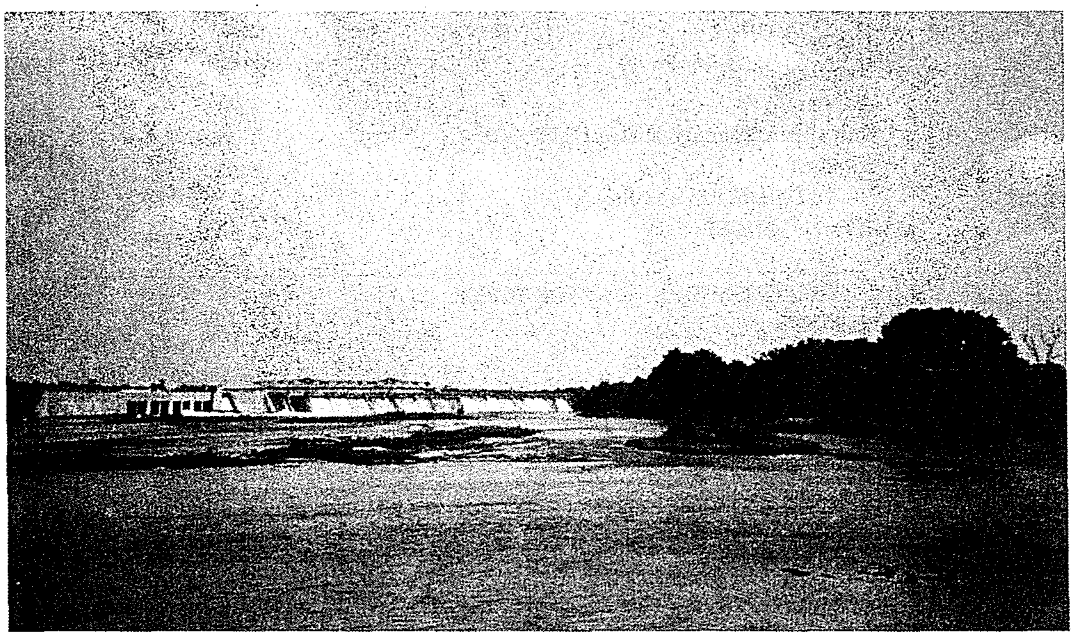

Figure 1.3 Habitat conditions in the Upper Dresden pool below Brandon Road Dam at the confluence of the river with Hickory Creek into which most of City of Joliet treated wastewater effluent and CSOs are discharged.

Table 1.1 Public wastewater treatment plants and their effluent flow on the Des Plaines River and Tributaries (average effluent flow greater than 1 cfs)

River	Wastewater (sewage) treatment plant	Average effluent flow (cfs)
Little Calumet River North and South	MWRDGC Calumet STP	290.00
Thorn Creek	Thorn Creek Sanitary District STP	15.00
Chicago River	MWRDGC Northside Chicago STP	367.00
	NSSD Clavey STP	15.20
	Deerfield STP	3.60
Chicago San. Ship Canal	MWRDGC Stickney STP	1,007.00
	MWRDGC Lemont STP	2.80
	Lockport STP	1.90
	TOTAL FROM CSSC	1,702.25
Des Plaines River Upstream of Brandon Pool		
	Lindenhurst STP	1.00
	NSSD Waukeegan STP	18.50
	NSSD Gurnee STP	16.20
	Libertyville STP	3.40
	Mundelein STP	3.70
	New Century STP	1.70
	Des Plaines STP	6.80
	MWRDGC Kirie STP	40.90
	Hindsdale STP	10.90
Salt Creek	MWRDGC Egan STP	24.60
	Roselle STP	1.70
	Bensenville STP	1.70
	Itasca STP	2.00
	Bensenville STP	1.70
	Adison STPs	8.90
	Salt Creek Sanitary District STP	2.00
	Elmhust	6.50
	Wood Dale North and South	4.8
Des Plaines River	Romeoville STP	1.50
	TOTAL FROM DES PLAINES RIVER	158.50
TOTAL TO BRANDON POOL		1,860.75
Dresden Island Pool		
From Brandon Pool		1,860.75
Hickory Creek	Frankfort STPs	1.83
	East Joliet STP	17.00
Des Plaines River	West Joliet	3.70
	TOTAL I-55 Bridge	1,883.28

In the reach just below the confluence of the Des Plaines River with the CSSC, the Section 303(d) list also identifies nutrient enrichment/low dissolved oxygen and flow alterations as parameters of concern. The UAA addresses these pollutants of concern, in addition to the proposal for a change of the current designated use.

Significant progress has been made in improving the water quality at the Stickney, Calumet, and other reclamation plants discharging into the Des Plaines River system. About 85% of the CSO discharges from the Chicago metropolitan area are now conveyed into the TARP system and receive treatment in the Stickney and Calumet plants. The lesser use of "secondary contact recreation and indigenous aquatic life" was applied in the 1970s .

The time has come to re-evaluate the designated use consistent with the goals of the Clean Water Act and to determine whether the higher use would be realistically attainable. Uses of the water body for navigation and wastewater and storm runoff disposal may be conflicting with the higher statutory designated uses (aquatic life protection and propagation and primary contact recreation) and relate directly to attainability of and influence the extent of aquatic life and contact recreation functions of the water body. It will be the task of this UAA to develop conditions for the higher uses and test them against reason 6 of the UAA which is the avoidance of widespread adverse socio-economic impact.

Table 1.2 Power plant design capacities and heat rejection (Holly and Bradley, 1994)

Station	Rated Load MW	Condenser Discharge cfs	7 day duration 10 years low flow, cfs	Heat rejection rate 10^6 btu/hr	$\Delta T°$ across the condenser °F	Summer $\Delta T°$ in the river (canal) at low flow*, °F
Fisk (one unit)	325	470		1288	12.2	
Crawfort (two units)	540	852		2243	11.7	
Will County (four units) CSSC	1095	2000		4982	11.1	8.7 (2550**)
Joliet (three units) Dresden Pool	1360	2620	1950	6417	9.4	6.7 (2850**) 8.93 (1950)

* The ΔT values are taken from the modeling study by Holly and Bradley and do not represent actual measured values and do not incorporate the effects of cooling towers. Twenty-four cooling towers were installed at the Joliet Station 29 that are used, as needed, to cool approximately 1/3 of the condenser cooling water flow from the Station.

** Low summer average daily discharge that is exceeded 90 percent of time based on 46 year simulation by Holly and Bradley.

Historic Development of the River

The Des Plaines River watershed and the investigated segment of the Lower Des Plaines River are located in the Central Cornbelt Plains ecoregion (Omernik, 1987). Historic annals from more than one hundred years ago described the Lower Des Plaines River at Lockport as a small stream. "*Its*

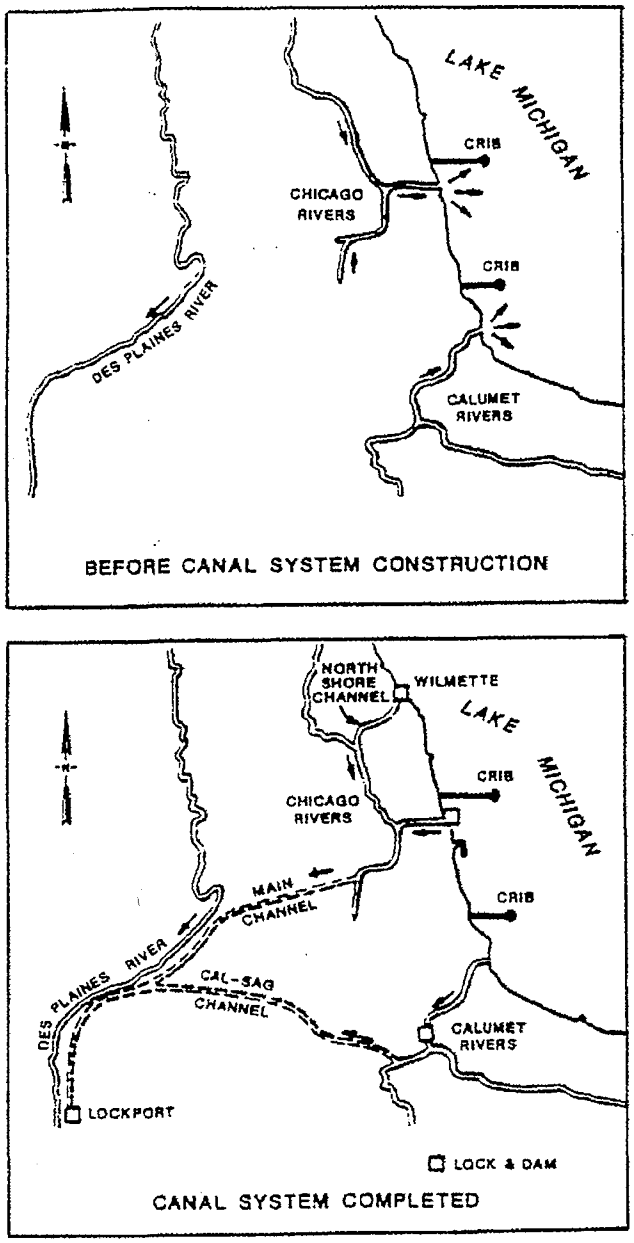

Figure 1.4 Upper Illinois Waterways before and after the construction of the CSSC (Source US Army Corps of Engineers; Macaitis et al., 1972)

normal water supply comes largely from marshy districts, but its flow is extremely variable, because of very rapid run-off in times of heavy rain or sudden thaws. Its waters are charged with organic matter from marshes and in later years its upper section receives considerable local sewage from suburbs of Chicago" (Palmer, 1903).

In the pre-development times at the beginning of the nineteenth century, Mud Lake, a part of the Des Plaines River and Chicago portage route paralleling today's I-55 upstream from Lockport, was *a large, leech-infested puddle filled with dense grasses* (Hill, 2000). The lake was essentially a marsh, one of many lining the Des Plaines River in these times. Mud in the lake was waste deep, thus, one could describe the lake in today's terms as having characteristics of an eutrophic to hypereutrophic water body, nearing the end of the geological eutrophication process that started during the ice age as a part of the prehistoric Lake Michigan outlet. At times of high flow, the Des Plaines River overflowed through Mud Lake easterly into the Chicago River. The river was described in 1821 as *".. present to the eye a smooth and sluggish current, bordered on each side by an exuberant growth of aquatic plants, in some places, reach nearly across the channel ... the water oftentimes filled with decomposed vegetation ... there is perhaps no stream in America whose current offers so little resistance in the ascent... "* (Elliott, 1998). In many places there were floodplain forests along the banks, some preserved even today.

The above discussion indicates that the water quality of the predevelopment Des Plaines River might have resembled the quality of wetland streams with occasional low dissolved oxygen (especially during night and early morning hours), and elevated levels of organics. Typically, wetland streams are dystrophic, meaning, that the nutrient levels and dissolved oxygen are low.

Conveyance of Chicago sewage into the Des Plaines River began in 1860 through the Illinois and Michigan canal. A pumping station with a capacity of 330 cfs was built at the junction of the canal with the South Chicago River. Apparently none or very little Lake Michigan water was pumped into this canal at that time. Between 1865 and 1871, the canal at the summit (subcontinental divide) was deepened to provide another 300 to 400 cfs of flow by gravity from the lake (Palmer, 1903). However, in a few years, sliding of banks and washing of silt into the canal diminished the gravity lake flow to less than 160 cfs. At the beginning of the twentieth century, Palmer (1903) noted that "*the city was growing rapidly in the last quarter of the nineteenth century and the slaughtering and manufacturing industries were enormously increasing, so that notwithstanding the diversion of part of the sewage into the canal, the river became even more and more offensive, and the people of the city suffered not only from the disagreeable and offensive character of the putrefying contents of immense stagnant cesspools or septic tanks situated in their midst...*"

The flow of polluted Chicago and Calumet Rivers into Lake Michigan had severe public health consequences. In the 1870s and 1880s, Chicago had the highest municipal typhoid rate in the United States (Macaitis et al, 1977). In 1889, the Illinois State Legislature created the Chicago Sanitary District to solve this acute health problem. The District is the predecessor of the Metropolitan Water Reclamation District of Greater Chicago (MWRDGC). As a solution to Chicago's problems with epidemics and unhealthy water quality of the Chicago River, in the second half of the nineteenth century, the Chicago Sanitary and Ship Canal (CSSC) was built (Figure 1.4) by the District

(MWRDGC). The operation of the canal reversed the flow direction of the Chicago River. The canal parallels the Des Plaines River and the old Illinois - Michigan Canal. It diverts Lake Michigan water into the Chicago River and further into the CSSC that connects the South Branch of the Chicago River with the Des Plaines River. To build the canal, 13 miles of the Des Plaines River were rerouted into a diversion channel in the late 1800s. The CSSC was finished at the beginning of the 20th century and navigation on the older Illinois - Michigan canal ceased in 1933.

Between 1907 and 1910, the District (MWRDGC) constructed a second sanitary canal called the North Shore Canal. This canal extends from Lake Michigan at Wilmette south 6.14 miles to the North Branch of the Chicago River and the flow continues to the CSSC. The Wilmette Controlling Works regulate the amount of Lake Michigan flow allowed into the canal and, ultimately, to the Des Plaines River.

The third canal, the Calumet Sag Canal, was completed in 1922. The canal connects Lake Michigan, through the Grand Calumet River, to the Sanitary and Ship Canal. This canal carries sewage from South Chicago (IL) and East Chicago (IN) to the CSSC and then to the Des Plaines River. The O'Brien Lock and Dam located on the Calumet River, regulates the flow of Lake Michigan waters into the canal. The Calumet-Sag Canal is 76 miles long and joins the main CSSC drainage canal at Sag, about 15 miles upstream from Joliet, Il.

Originally, the development of the Lake Michigan diversion project by CSSC, North Shore and Calumet - Sag canals were undertaken and justified by the state of Illinois that the state would make a profit by providing water energy. No diversion was needed to provide a connecting navigable waterway, as distinguished from the requirement for providing conveyance of sewage from the Chicago metropolitan area to the Des Plaines and Illinois Rivers, instead of into Lake Michigan. However, the large diversion of water from Lake Michigan at the early time of the CSSD was made by the state of Illinois without the consent of any of the states bordering the Great Lakes. Temporary permits were from time to time granted by the Secretary of War solely on the request of the Chicago Sanitary District and the state of Illinois on the grounds that a termination or reduction of the diversion would impair the health of the people in Chicago (Naujoks, 1946). Originally, Secretary of War issued a permit authorizing a diversion of 4,167 cfs. However, it took more than 25 years until (in 1925) the Supreme Court entered a decree allowing the Secretary of War to issue the diversion permits. In March, 1925, the permit issued limited the diversion to 8,500 cfs.

In 1922, 1925, and 1926, several Great Lakes states filed court actions in the US Supreme Court seeking to restrict the diversion into the CSSC and Des Plaines River from Lake Michigan in Chicago. A Special Master, appointed by the US Supreme Court to combine the three suits and hear the case, found in 1925 that the permit was valid and recommended dismissal of the action. However, the U.S. Supreme Court reversed the Special Master's findings and the Court instructed the Special Master to determine steps necessary for Illinois and MWRDGC to reduce the allowable diversions. Consequently, a 1930 decree reduced the allowable diversion in three steps: to 6,500 after July 1, 1930; to 5,000 cfs after December 1935; and to 1,500 after December 1938 (Naujoks, 1946). The diversion is water from Lake Michigan and does not include domestic pumpage.

In 1975, the discretionary diversion of flows into the Lower Des Plaines River was as follows (Macaitis et al., 1977):

Domestic pumpage	1,658 cfs
Stormwater runoff	977 cfs
Lockages and leakages	226 cfs
Water required for maintenance of navigation	58 cfs
Total	2,919 cfs

The CSSC fully reversed the flow of the Chicago River and is currently bringing a total of 3,200 cfs of lake water into the Des Plaines River. Actual diversions may be less. The 3,200 diversion is divided between the flow augmentation and sewage resulting from the use of the allotted lake water diversion for domestic and other water supplies. Of the 3,200 cfs, approximately 2,400 to 2,600 cfs is the actual lake diversion that enters the CSSC as (a) wastewater, (b) lake flow for water quality purposes (dilution) and navigation, and c) 600 to 800 cfs is runoff water diverted from the lake Michigan watershed into the Chicago River and the CSSC.

The annual average "clean" lake flow water allowed for diversion into the waterway is only about 320 cfs; however, apparently this flow can be released primarily during the summer low flow periods at a higher prorated rate.

The flow reversal has resolved the public health problem and the pollution of Lake Michigan, the main source of potable water for the city and its suburbs, but also diverted the pollution into the Des Plaines River. In 1911, observations by two biologists noted and reported septic conditions for twenty-six miles of the Illinois River from its origin (confluence of the Des Plaines and Kankakee rivers) and the Des Plaines River downstream from Joliet (Mills et al., 1966).

Significant improvements of water quality were achieved in the last century by building and implementing secondary treatment at the large treatment plants operated by the Metropolitan Water Reclamation District of Greater Chicago in the North Shore, Calumet and Stickney, by smaller MWRDCG and suburban community secondary treatment plants, and by implementing industrial treatment of wastewater required by Sections 301(b)(1)(A) and (B) and 306 of the Clean Water Act. The Stickney plant is the largest in the world.

The tunnel and reservoir project (TARP) is designed to eliminate overflows from the combined sewers into the Chicago River and further from the CSSC waterway. The tunnel was put, leg by leg, into operation since 1985 (the main leg of the mainstream tunnel was partially in place in May 1985 and fully operational in October 1985). Today, the tunnel part has been mostly implemented. The overflow water (mixed with some groundwater inflow into the tunnel) is stored in the tunnel and pumped to the Stickney and Calumet plants for treatment. A 10.5 billion gallon reservoir is being built near the pumping station near McCook and another reservoir will be built in the northern section of Thornton Quarry. When the reservoirs are on line (approximately in 2010), the combined sewer overflows and back flow of the Chicago River into Lake Michigan during wet weather will be greatly reduced and all dry weather and wet weather waste flows will be treated prior to discharging into the CSSC and, subsequently into the Lower Des Plaines River.

Another step that changed water quality in the CSSC and Des Plaines River was the elimination of chlorination of the treatment plant effluents in 1983 and 1984. Although the effluent chlorination reduced bacteria in the effluent, the residual chlorine was toxic to the aquatic life. The effect of chlorination on bacteria densities in the Des Plaines River will be discussed in more detail in Chapter 7 of this UAA.

Lastly, it appears that several years ago, a change in plant aeration and operation has resulted in dramatic decrease of ammonia levels in the effluent and the entire system of the CSSC and Lower Des Plaines River.

Today, at least 25 fish species, including white crappie, large and small mouth bass, green sunfish, bullheads, and many minnows, are now found regularly in the CSSC and Des Plaines River system (Hill, 2000).

The Lower Des Plaines River today is a highly modified and managed riverine system. The changes are irreversible in the long run and the system cannot be returned to the predevelopment conditions nor to some kind of natural stream. The Use Attainability Analysis must consider this status and find the best ecological use of the water body also considering its other uses for navigation, waste disposal and cooling. In order to meet its ecological goals, the system will require extensive management and the users must also be aware of limitations imposed on their use by other demands on the river.

History of Use Designation and Water Quality Standards in Illinois[1]

The state of Illinois currently recognizes two designated uses of the state's navigable water bodies:

I *The General Water Use*, and
II *Secondary Contact and Indigenous Aquatic Life Use Designation*

The General Use conforms with the Clean Water Act Section 101(a) goals, and the corresponding standards are in accordance with or even more stringent than the federal criteria (USEPA, 1986 and subsequent documents).

The Secondary Contact and Indigenous Aquatic Life is contained in Sections 303.204 of the Illinois Pollution Control Regulations (Ill. Adm. Code Title 35). It is described as

> *"...those waters not suited for general use activities (fishing, swimming, aquatic life protection, agricultural and industrial uses, etc.) but which will be appropriate for a secondary contact use*

[1]Portions of this section are taken from an IEPA document describing the use designations.

and which will be capable of supporting an indigenous aquatic life limited only by the physical configuration of the body of water, characteristics and origin of the water and the presence of contaminants in amounts that do not exceed the water quality standards..." (35 Ill. Adm. Code 302.402).

The following water bodies have been approved for the Secondary Contact use designation in northeastern Illinois (Illinois Pollution Control Board Rules and Regulations-Chapter 3: Water Pollution):

- The Chicago Sanitary and Ship Canal
- The Grand Calumet River
- The Calumet River, except the 6.8 mile segment extending from the O'Brien Lock and dam to lake Michigan
- The Calumet - Sag Channel
- Lake Calumet
- The Little Calumet River from its junction with the Grand Calumet River to the Calumet - Sag channel
- The Calumet River
- The South Branch of the Chicago River
- The North Branch of Chicago River
- *The Des Plaines River from its Confluence with the CSSC to the Interstate 55 bridge*
- The North Shore Channel

Development and Adoption of the Secondary Use and Indigenous Aquatic Life Use

Prior to adoption of the Illinois Environmental Protection Act in 1970, water quality management activities, including establishment of water quality standards, were under the jurisdiction of the Illinois Sanitary Water Board. Pursuant to the federal Water Quality Act of 1965 (PL89-235), the Sanitary Water Board initially designated the Lower Des Plaines River as an "Industrial Water Supply Sector" with numeric and narrative criteria appropriate to such use category. Stream uses specified within this classification included "commercial vessel and barge shipping, recreational boating transit, withdrawal and return of industrial cooling and process water, and to receive effluents from industrial and domestic waste treatment facilities." Narrative standards established minimum conditions such as freedom from bottom deposits, floating debris, nuisance, and toxic conditions. Water quality standards for dissolved oxygen, pH, temperature, dissolved solids, and bacteria were also included in Rule 1.07 of SWB-8 which was adopted by the Sanitary Water Board on December 1, 1966. Following adoption of the initial water quality criteria, the Sanitary Water Board submitted a plan for implementation of the standards applicable to the lower Des Plaines River to the federal government on August 10, 1967.

Upon enactment of the Illinois Environmental Protection Act in 1970, the Sanitary Water Board was superseded with creation of the Illinois Pollution Control Board (Board) and the Illinois Environmental Protection Agency (Agency). While Sanitary Water Board regulations remained in place on an interim basis under the new state statute; the Board and Agency focused attention almost immediately on development of new water quality standards. Draft proposed rules were published for public comment on May 12, 1971 (docketed as R71-14) and public hearings were conducted shortly thereafter.

At the September 14, 1971 public hearing in Joliet, the previous standards were discussed along with the proposed revisions. At the time of the hearings, the Board was proposing that the Chicago Sanitary and Ship Canal be classified as restricted upstream of its point of contact with the Des Plaines River, generally recognized as located at Lockport. Downstream from Lockport, the Board proposed to change the river's designation to the more stringent general use. Restricted use standards were provided for waters that were not protected for aquatic life and in which aquatic life standards for various toxic materials need not be met (similar to the industrial water supply use designation under the SWB regulations). Restricted use later became known as the "secondary contact and indigenous aquatic life" use. The significant changes in the proposal involved the waters that were previously designated as industrial water supply use and had to meet the primary contact, general use standard (this includes the Des Plaines River).

The Commonwealth Edison power company immediately suggested that the restricted use designation be extended to include the Des Plaines River down to the point of the Interstate 55 bridge. Others giving an opinion on this issue included Richard Ciesla, Director of Utilities for the City of Joliet. Mr. Ciesla's concern was that the City of Joliet, being downstream of the proposed use change at Lockport, would be forced to comply with the more stringent general use standard even though the waters had not come to a point of dilution. He suggested the point of changeover be made at the confluence of the Des Plaines and the Kankakee Rivers (IPCB Hearing, Sept. 14, 1971). The United States Steel Corporation of Joliet was also concerned that the Board had overlooked the fact that the area south of the proposed change was industrial and suggested that the restricted use be extended a short distance to the area near Brandon Locks (letter, November 9, 1971). Another concerned organization from Joliet was the Will-Grundy Manufacturers' Association, who suggested that the restricted use designation be "extended south at least to a point where industrial land is not a consideration" (letter, November 9, 1971).

Another Board hearing was held on February 10, 1972, at which Commonwealth Edison provided a panel of witnesses to support their opinions of the water quality standard. The witnesses concluded that the costs of imposing a general use water quality standard on the Des Plaines would far outweigh any benefits. Also, according to the witnesses, even if water quality standards could be met, the river upstream of the I-55 bridge would not be suitable for aquatic life due to heavy industrialization, barge traffic, diking of the shoreline and dredging.

Meeting the general use standard for temperature was the greatest concern for Edison. Witnesses were doubtful of the possibility that general use temperature standards could be met until the Des Plaines' confluence with the Kankakee (five miles from the I-55 bridge). Arguments were also made suggesting that meeting the temperature standard was not important due to the small possibility that the general use water quality standards would be met in other aspects. Therefore, while an increased temperature standard had perceived benefits such as maintaining the river for year-round navigation and speeding up the degradation of ammonia, there would be no advantage in adopting a general use designation because the waterway would be incapable of supporting aquatic life anyway and use of the river for recreation up to the I-55 bridge was nonexistent due to industrialization. In the non-industrialized five-mile stretch; however, support for aquatic life needed to be addressed. The fish biologist, called as a witness for Commonwealth Edison, testified that fish would rarely be disturbed by an increased temperature standard, and on the occasions when the temperature did raise above tolerance levels, the fish would sense the rise and simply move out to other waterways until the temperature was once again suitable.

Cost of Cooling Towers was an Overriding Issue

The Opinion of the Board dated March 7, 1972 addressed the issues that were raised by Edison's witnesses. Page ten, Part II (205) discusses restricted use standards and states "The temperature standard has been modified in response to a suggestion from Commonwealth Edison Company, in order to avoid expensive cooling devices that are not necessary to the avoidance of nuisances or safety hazards." In Part III the restricted use designation is discussed and the section of the Des Plaines adjacent to the Chicago River System is included in the category. Once again, the expense of cooling towers was noted and the Board stated that meeting temperature standards for aquatic life would be futile in an area where standards could not be met for dissolved oxygen (and perhaps ammonia). The Board's decision, therefore, was to classify the Des Plaines River from Lockport to the I-55 bridge as restricted use waters.

During the hearings, a representative of the USEPA testified in general support of the restricted use designation and the waters that carried that designation. The problem identified centered primarily around semantics and consistency with federal guidelines.

Finally, the November 8, 1973 Board Opinion discusses the I-55 boundary on the Des Plaines at page five. In the opinion, it is stated that "The basis for the Board's decision to use the I-55 bridge as a boundary for the division of the Des Plaines River into restrictive and general use is that the location of the bridge corresponds to changes in the physical environment characteristics of the area." The industrial characteristics described by Edison's witnesses in reference to the Des Plaines could not be applied to the area below the bridge. The Board also found the five-mile stretch, downstream of the I-55 Bridge, "is capable of providing sources of recreating badly needed in the area (R. 107, 9/14/72), and is supporting a limited desirable aquatic biota." The November 8, 1973, Opinion of the Board can be found in Appendix A.

In the same opinion, the Board also addressed the dissolved oxygen and thermal standards. The Board urged the Metropolitan Sanitary District of Greater Chicago to give serious considerations to such further measures, including in-stream aeration, that offers promise of improving the quality of its restricted use waters. It modified its original requirement to reduce the effluent BOD_5 to 4 mg/L and allowed MWRGC to reduce BOD_5 in its effluents to 10 mg/L and to prove to the agency (IEPA) by the end of 1977 that this effluent BOD concentration would meet the DO standard. Two prominent experts testified that the standards could be met by both restriction of BOD_5 and instream aeration.

In its November 8, 1973 Opinion (Appendix A), the Illinois Pollution Control Board proposed to amend Section 302 Restricted Use Waters by adding a clause requiring the Board to hold hearings in 1973 and every five years thereafter to determine whether any Restricted Use Water should be reclassified as a General Use Water. *This amendment was in response to the Federal Environmental Protection Agency policy not to approve restricted status as a permanent status for any water.* After holding several hearings in 1973, Board modified the Restricted Use designation so it was consistent with federal requirements. The change renamed the designation "Secondary Contact and Indigenous Aquatic Life" and incorporated the concept of protecting attainable uses including aquatic life that were limited only by the physical constraints of the waterway. Since the adoption of the order (IPCB Docket #73-1), the language of the designation, and most numerical standards have remained substantially unchanged. The magnitude of the Illinois General Use and Secondary Contact Use and corresponding federal USEPA criteria are presented in Chapter 2.

From the above description of the history, it is clear that the secondary contact/indigenous aquatic life use had its origin before the enactment of the Clean Water Act. It was based primarily on the feeling of hopelessness for any substantial improvement of the water quality of the river on the part of the agencies that were prevalent at the beginning of 1970s and on economic reasons to accommodate effluent and heated discharges into the river that was deemed incapable to support aquatic life and provide for recreation.

Description of the Secondary Contact and Indigenous Aquatic Life Designation

There is one basic underlying common characteristic of the waterbodies that have been included into the Secondary Use Contact and Indigenous Aquatic Life designation in northeastern Illinois: these water bodies were a part of a massive engineering effort that reversed the flow of the Chicago River System and the Upper Illinois Waterway to allow the City of Chicago to divert its wastewater from Lake Michigan. Although the original official justification for creating the Chicago Waterway System and the flow reversal was presented differently, there is no doubt that the system had a tremendous beneficial impact on public health. The IEPA document stated that at the time the Secondary Contact Use (1970s) was formulated, the waters designated for this use had the following common characteristics:

1. Heavily dredged and maintained channel including steep-sided cross-sections designed to accommodate barge traffic with minimal clearance, and/or optimize flow.
2. Significant sludge deposition which is the result of combined sewer overflows and urban runoff. Sludge depth in the channel system can reach five feet or more despite dredging.
3. Flow reversal projects, such as this one, place a premium on head differential. The entire system has minimum slope and, consequently, low velocity, stagnant flow conditions. Because of the need to minimize use of Lake Michigan water, diversion to maintain flow in the system is kept as low as possible.
4. Urban stress is significant within the entire drainage area. There was essentially no recreation potential with most adjacent property commercially owned and access limited.

5. Habitat for aquatic communities in the main channel was nonexistent due to the impact of commercial and recreational watercraft use of the system as well as sludge deposition. Watercraft lockage through the Chicago River Control Works averages 25,000 vessels annually; most activity occurs during the summer months.
6. In addition to the above man-made and irretrievable modifications to the Chicago River System that are designated as Secondary Contact use, the system also carries a massive wastewater load. During winter periods, dry weather flow is 100% wastewater. During summer periods, a small "discretionary diversion" of Lake Michigan water is permitted to minimize the combined effects of waste loads from the municipal and industrial discharges to the system and poor assimilative capacity. During wet weather periods, flow in the system is made of wastewater and combined sewer overflows.[2]

In the period of twenty years following the use designation in 1972, the agencies struggled to find the potential ecological use of the Chicago Waterways. Twenty to thirty years ago, water quality was bad and appeared to be getting worse. Table 1.3 reports the DO concentrations taken from an extensive study of the Upper Illinois River Waterway by Butts et al. (1975).

However, the study also documented a beneficial impact of dams on the DO concentrations and reported a compliance with the DO standard in the Upper Dresden Island pool below the Brandon Road Dam. The Lockport Dam and power house operation increased the DO concentrations between upstream and downstream of the dam by about 1 mg/L while the Brandon Road dam overflow (Figure 1.3) increased the DO content by almost 5 mg/L. It should be noted that the aeration efficiency of dams increases with the oxygen deficit. The re-aeration at Lockport was intermittent because at lower flows all flows were diverted to the powerhouse. Butts et al concluded, after an extensive modeling study, a DO standard greater than 3.0 mg/L was not realistically achievable at the time of the study (1970s).

Table 1.3 Historic (1970s) Concentrations of the Dissolved Oxygen (Butts et al, 1975)

	DO concentration (mg/L)		
Location	Max	Average	Minimum
Brandon pool			
- upstream	2.7	2.0	1.1
- downstream (above the dam)	1.5	1.1	0.6
Dresden Island			
- below Brandon Dam	6.6	5.9	5.4

[2] The above six items describe the understanding of the system in the 1970s. Thirty years later the situation in the Lower Des Plaines River has significantly improved. Although the hydrologic conditions of the flow and diversions remain about the same, water and sediment quality has improved. Also, the habitat that was characterized as nonexistent in the 1970s has improved, especially in the Dresden Island pool. The assessment of current water quality and habitat conditions is presented in Chapters 2-6.

In the 1970s, the macroinvertebrate composition at most stations was limited to sludgeworms and bloodworms. The number of worms in the samples above the mile 281.4 (Dresden Island pool) was so great that field picking and counting was impossible (they existed in hundreds of thousands per square meter). The sediment oxygen demand (SOD) in Brandon pool was measured ranging from 40 to 50 g/m^2-day, an unusually and unsustainably high rate[3], but SOD in Dresden Island pool between miles 283 and 286 was only 1.1 to 2.7 g/m^2-day. Fecal coliform densities were very high, exceeding current levels by two orders of magnitude.

In 1972 Congress passed the Clean Water Act Amendments to the Water Pollution Control Act. In the same year, the Illinois Pollution Control Board was formulating the uses of the Illinois water bodies and the appropriate standards to protect these uses (Illinois Pollution Control Board, March 7, 1972). In this rule, the IPCB redefined the General and Restricted Uses. It ruled *"that all waters should be protected against nuisance and against health hazard to those near them; that all waters with exception of a few highly industrialized streams consisting primarily of effluents in the Chicago area, should be protected to support such life..... Consequently general standards for water quality are set that will protect most uses except public water supply; .... and more lenient standards are set for those streams classified for restricted use."*

Establishment of the "restricted use," later renamed "Secondary Contact and Indigenous Aquatic Life" use, was limited to "*those waters in the Chicago industrial area for which it was established, that even with the most advanced treatment and with stormwater overflow control, aquatic life standards (for dissolved oxygen and perhaps for ammonia) cannot be met ... and that meeting the aquatic temperature standards in the same areas, as well as in adjacent sections of the Des Plaines River, would require cooling towers costing tens of millions of dollars and produce doubtful benefits in terms of stream improvements*".[4]

In the 1980s the USEPA re-evaluated the appropriateness of Secondary Contact and Indigenous Aquatic Life designation for the Chicago waterways, including the Lower Des Plaines River (an memorandum by Jim Park to IEPA and provided to AquaNova/Hey Associates team). The USEPA concluded in the mid 1980s that the waterways designation for secondary contact use in Illinois was appropriate, in spite of the fact that no Use Attainability Analysis was submitted. The USEPA agreed with the IPCB that the primary contact activities were also inappropriate for these waters due to limited access and danger associated with heavy navigation as well as general aesthetic constraints. The USEPA apparently, in mid 1980s, approved elimination of the bacterial water quality standards for secondary contact waters and supported elimination of this use.

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[3] Recent research findings identified ebullition of methane and ammonia from sediments and their oxidation in the upper sediment layer as the primary cause of SOD. The SOD is limited by the rates of methane oxidation and ammonia nitrification and its maximum rate is about 6 g of O_2/m^2-day (DiToro, 2000; DiToro et al., 1990; Novotny, 2002).

[4] In the early 1970's, cooling towers were not common and were expensive. Today cooling water technology using forced and natural draft is commonly used by and mandatory for many power plants on rivers that have a similar size as those located on the Des Plaines River, e.g., plants operated by the Tennessee Valley Authority or by Wisconsin Energies on the Wisconsin River and Kenosha, WI.

The current situation of water and sediment quality and the status of the attainment of the General and the Secondary Contact and Indigenous Aquatic Life uses will be extensively documented and discussed in the subsequent chapters. More than thirty years after the Secondary Contact and Indigenous Aquatic Life Use has been instituted by the IPCB and IEPA, the time has come to re-evaluate the current situation (existing use) and consider, if appropriate, a use that would either meet or approach the statutory uses required by Section 101(a) of the Clean Water Act.

Organization of this Report

This study begins with the definition of the general use and follows with the assessment of the compliance or noncompliance with the general use standards. For those compounds that do not meet the standards, the study looks for reasons of noncompliance and attainability. For pathogens, the study applies the USEPA bacterial criteria that use Escherichia Coli as indicator organisms. Ecologic evaluation and criteria were used to define the ecologic potential of the two investigated segments. A new site specific use was then defined for the Brandon Road pool.

This Use Attainability Analysis report is organized into nine chapters:

1. **Introduction** (this chapter)

2. **Water Body Assessment - Chemical Parameters**
This chapter describes the methodology used for water body assessment, current standards and current water quality as described by 25 chemical parameters. It divides the parameters into those that are in full compliance with the general use standards and those that are not. A more detailed analysis of noncomplying parameters follows.

3. **Water Body Assessment - Sediments**
Significant improvements in water quality were followed by improvement in sediment quality. The sediment quality was characterized by the Illinois comparative criteria and, in some cases, by calculating the pore water concentrations.

4. **Water Body Assessment - Physical Assessment**
This chapter evaluates the physical attributes of the Brandon Road and Dresden Island pools and their habitats.

5. **Evaluation of Existing and Potential Macroinvertebrate Community**
Enumeration and evaluation of indices of biotic integrity is a cornerstone for assessment of the ecologic potential of the river. Macroinvertabrate communities are used as an indicator of ecological health.

6. **Evaluation of Existing and Potential Fishery Community**
Fish community structure has long been used as an indicator of ecological stress. Numerous reference water bodies were selected and analyzed for the impact on biotic integrity of navigation, impoundment and pollution.

7. **Pathogens and Recreation**
This extensive chapter evaluates the current water quality expressed by the fecal coliform indicator organisms and attainability of the federal criteria that use Escherichia Coli and enterococci as indicators. The federal criteria add flexibility regarding the selection of the risk to which the magnitude of the standard could be related. The chapter specifies options for site specific recreational uses for the Brandon Road and Dresden Island pools.

8. **Modified Water Use Designation for Brandon Road Pool and Use Upgrade for the Lower Des Plaines River**

This chapter defines the general use for the Dresden Island Pool and a site-specific modified use designations for the Brandon Road Pool with corresponding standards.

9. **Suggested Action Plan**
Actions needed to accomplish the goals specified by this UAA are outlined.

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CHAPTER 2

WATER BODY ASSESSMENT: CHEMICAL PARAMETERS

Introduction

This chapter presents the water body assessment of the chemical integrity for the Lower Des Plaines River from its confluence with the Chicago Sanitary and Ship Canal to the I-55 Bridge (Figure 1.1). This assessment is an integral part and the first step (Figure 2.1) of the Use Attainability Analysis for the Lower Des Plaines River that screens the available chemical sampling data to determine which parameters are currently meeting the State of Illinois General Use Water Quality Standards and which are not. The parameters that do not meet the standards, or if there is a threat that they may not meet the standards in the near future (one or two reporting cycles), are then further analyzed. The attainability of the designated statutory uses of fish and wild life protection and propagation, contact recreation and of the Illinois General Use Water Quality Standards are assessed. Chemical data analyzed in the report were provided by the following agencies:

- Illinois Environmental Protection Agency (IEPA)
- U S Geological Survey (USGS)
- Metropolitan Water Reclamation District of Greater Chicago (MWRDGC)
- Commonwealth Edison Company
- Midwest Generation, EME, LLC

Water Quality Criteria and Standards

The Use Attainability Analysis (UAA) provides a mechanism for change of the use or standards if a designated higher use (commensurate with Section 101(a) of the CWA) is not attainable. Also, if a lesser use was designated previously, the regulations require a UAA reevaluation and possible upgrade. The UAA has three parts (Figure 1.1) (Novotny et al., 1997): (1) Water Body Assessment (WBA), (2) Total Maximum Daily Load (TMDL) analysis, and (3) Socio-economic analysis. Most UAA problems are resolved by the first component, which is also the case of this UAA. This report represents the outcome of the WBA for the Lower Des Plaines River.

The use evaluation and analysis are accomplished by comparing the existing or future water quality to a set of water quality standards or criteria, followed by scientific assessment to find out whether the standards are attainable. Although several definitions of the term "standard" and "criterion" have been suggested in the literature (see Krenkel and Novotny, 1980), in this document we will use the term "criteria" for the USEPA defined limiting values (40 CFR 131) and "standard" for Section 302 binding limiting values established by the state of Illinois.

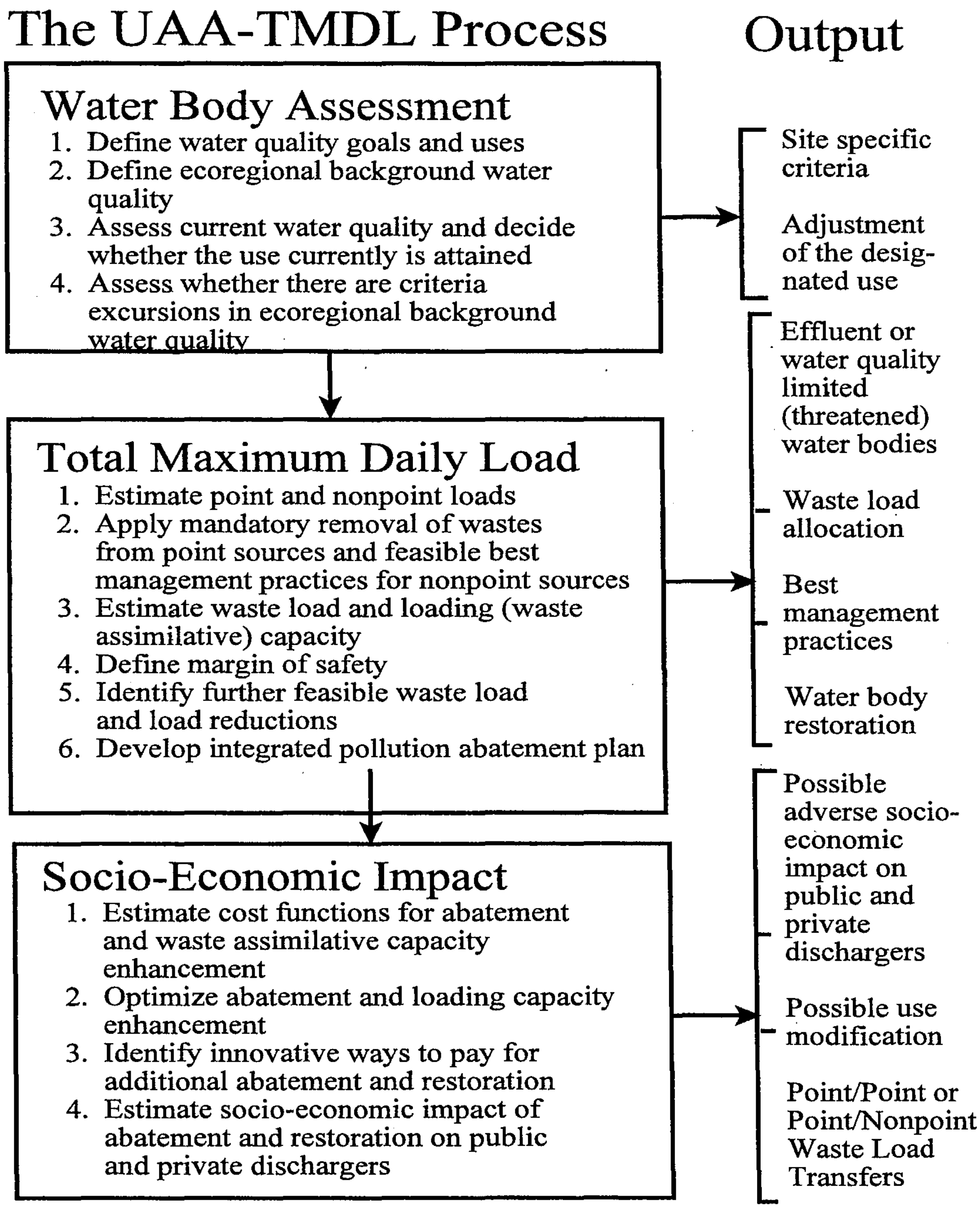


Figure 2.1 **Components of the Complete UAA Process. Water body assessment is the first component.**

Application of the Standards - Aquatic Life Protection

Generally, a standard (criterion) for a pollutant has three components (USEPA, 1994):

- Magnitude - How much of a pollutant (or a pollutant parameter such as toxicity), expressed as concentration, is allowable.

- Duration - The period of time (averaging period) over which the in-stream concentration is averaged for comparison with standard concentrations. The specification limits the duration of concentration above the criteria.

- Frequency - How often the standards can be exceeded.

Establishing these three dimensions of the water quality standards is crucial for a successful UAA and, by the same reasoning, for Total Maximum Daily Load (TMDL) studies (Committee to Assess the Scientific Basis of the TMDL Approach to Water Pollution Reduction; 2001). A subsequent modified TMDL will address the attainability issues for those few parameters that do not meet the general use (aquatic life protection and propagation and contact recreation) designation. The modified TMDL will be preceded by assessment of the impact of other possible causes of impairment listed as reasons 1 to 5 in Box 1.1.

Many states simplified the frequency/duration component by substituting the rule that *a numeric standard must be maintained* (not to be exceeded) *at all times*. Such limitation is statistical impossibility because there is always a chance - albeit very remote - that a water parameter may reach a high, but statistically possible, value exceeding an established standard (Committee to Assess the Scientific Basis of the TMDL Approach to Water Pollution Reduction; 2001). This requirement also brings ambiguity. For example, Figure 2.2 shows that it is possible if nine samples are taken over a period of three years, none of the samples could, by chance, result in an excursion. If a hundred samples are taken in the same period, one or a few (e.g., five or less) may exceed the standard. Statistically, these two situations are identical but the former would not result in violation while the latter would. Stream concentrations represent a statistical time series for which only infinitesimally large values of a standard would have a 100% statistical probability of not being exceeded *at all times*.

The procedure of probabilistic fitting/analysis has been used in hydrology and water quality analysis for many years. It has been described in almost every textbook on hydrology. It has been used during the USEPA evaluation of stormwater runoff during the National Urban Runoff Project (USEPA, 1983), by USGS in evaluation of the Upper Illinois Waterway (Terrio, 1990;1994), and long earlier works by the Illinois Water Survey (Butts et al., 1974). Use of statistics is indispensable in water quality reports and evaluations and should not be challenged. The log-normal statistical analysis methodology requires arranging measured values, transformed to their logarithms or plotted on a logarithmic scale, according to their order of magnitude (ascending for being ≤ [less or equal] used for most parameters and descending for ≥ used for dissolved oxygen) and assigning a probability

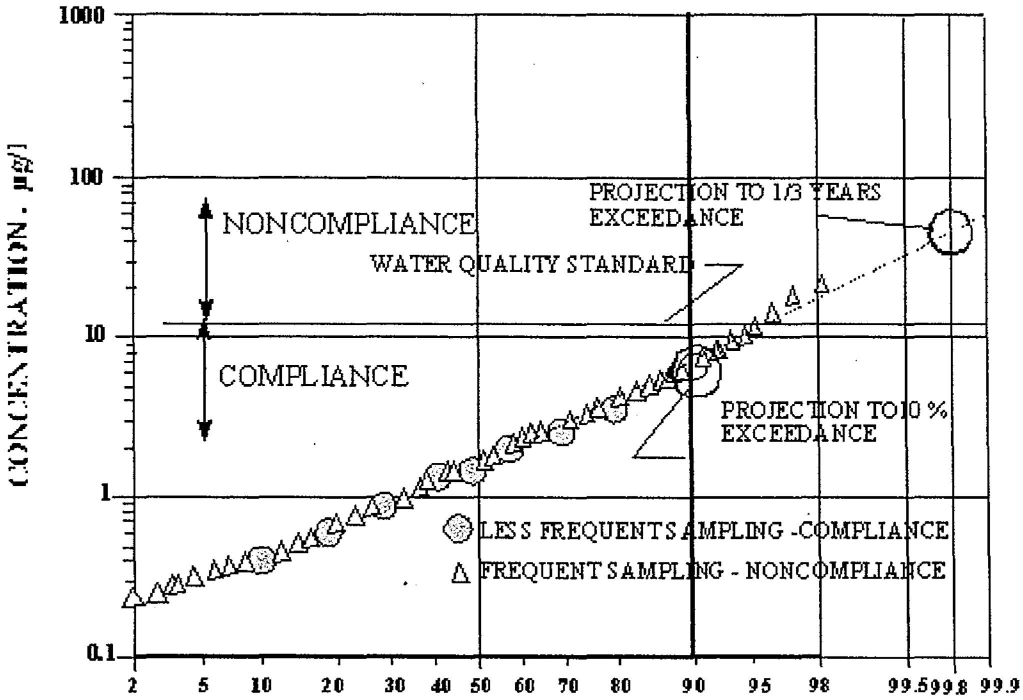


Figure 2.2 **Statistical Plotting of Data and Decision on Compliance with Standards (Designated Use). Compliance or noncompliance is revealed from the interceptor of the line of the best fit with the 99.8 percentile line (for 1 B 3 - once in 3 years allowable excursions) or 90% for 10% allowable excursions.**

plotting position as $p(\%) = \frac{100\,M}{N+1}$, where M is the order of magnitude and N is the total number of samples. Several commercial software packages are available for this type of analysis. Log-normal probabilistic plotting (Figure 2.2) is also used for convenience and presentations. If the data followed the log-normal probability distribution, the plot would result in a straight line; however, other probability distributions may also be used.

The plot and statistics behind it also prove that there is no such thing as "*compliance at all times*" because a value that would never be exceeded is in infinity (i.e., there is no 100 % ordinate on the plot). Plotting and analyzing the data on the probability plot provides a powerful visual tool for understanding the variability of the data and puts the smaller monitoring sample data on par with a sample with more measurement. Other aspects of this technique will be discussed in a subsequent section.

The federal criteria defined the permissible frequency of excursions for federal toxicity (priority pollutants) criteria. The Water Quality Standard Regulation (USEPA, 1992; 1994) specifies:

- *acute toxicity criteria* - 1 hr average concentration (essentially a grab sample) not to be exceeded more than once in 3 years on an average (1B3 allowable excursion)
- *chronic toxicity criteria* - 4 day average concentration not to be exceeded more than once in 3 years on an average (4B3) used for most toxic pollutants, or 30 day average concentration (30B3) that is used for ammonium toxicity

The USEPA selected the 3-year average frequency for criteria's excursions for priority pollutants with the intent of providing for ecological recovery from a variety of severe stresses. The 3-year recurrence was derived from observations on the length of recovery of ecosystems after a toxic spill. This return interval is roughly equivalent to the recurrence of 7Q10 design low flow conditions used for point sources. It should also be pointed out that even when the concentration of the constituent reaches the magnitude of the standard, the damage to the ecosystem may not occur because of the safety risk factors (margin of safety) incorporated into the magnitude value of the standard (USEPA, 1991a).

A frequency of once in 3 years of allowable excursions corresponds to a probability of 1/(365x3) = 0.001 or 0.1% of being exceeded or 0.2 % of being equaled or exceeded. Then 100 - 0.2 = 99.8 % should be the probability of compliance. Therefore, the critical decision point should be placed at the 99.8 % probability of being less for the acute (CMC) standard. Since most of the water quality constituent concentrations from a sufficiently long record follow log-normal distribution, the acute toxicity criterion (standard) would be violated if the 99.8 percentile of *maximum daily concentrations* arranged in the ascending order of magnitude on the log-cumulative probability plot would exceed the standard. One hour average values for acute toxicity would imply grab samples taken on randomly selected days or daily. For dissolved oxygen concentrations, the data could be arranged and plotted in a descending order of magnitude.

For chronic toxicity, the USEPA water quality guidelines (USEPA, 1992, 1994) require 4 days averaging (30 days for ammonium) periods. This would imply that samples must be taken daily or composited over a 4 day period. Such sampling programs are not available for the study area. Illinois interpretation and water quality standards allow averaging four consecutive samples that may not be one day apart. Some parameters (e.g., dissolved oxygen and temperature) have been continuously monitored; however, most stream water quality monitoring parameters are available only on a monthly or longer basis. For such an "incomplete" monitoring series that does not allow 4 day (30 day) averaging the USEPA, in one of their interim documents (Delos, 1990) after a rigorous mathematical analysis supported by analysis of continuous data, suggested that the chronic criterion (standard) should be applied to a 98.8 - 99.9 percentiles of data but this suggestion was not included in the criteria regulation. For a first cut assessment, the most realistic 99.4 percentile from the Delos' analysis, supported by monitored data on the Ohio River, is used to define the chronic standard in this report. Theoretically, this statistical value should be very close to that obtained by the Illinois interpretation of four days averaging. The water quality regulations do not allow excluding the chronic criteria (standards) because of unavailability of a "complete" daily time series of data. The 99.4 percentile value will be accepted with caution. For some pollutants that will require a more detailed analysis, specifically ammonium, a Monte Carlo Modeling is used to find four and 30 day moving averages of the data represented by incomplete series of observations.

For nonpriority pollutants, scientific judgement will be used for determining the frequency and duration components if not specified in the standard or criteria documents. In most cases, the duration component is specified (e.g., the magnitude of a DO standard or temperature can be exceeded for a specified number of hours) but the frequency component may be missing. In such cases, compliance with a standard will occur if:

- all measured data are below the standard, and/or
- 95 to 99 percentile of the data is below the standard

Table 2.1 contains the numeric Illinois General Use and Secondary Recreation and Indigenous Aquatic Life Standards and corresponding federal criteria. Table 2.2 presents a comparison of the narrative Illinois standards and federal criteria. Many of the standards and criteria are site specific such as metals and ammonium.

Water Effect Ratio (WER)

To overcome the problem of the toxicity difference between the total concentrations of potentially toxic compounds and their toxic fraction and toxicity, a parameter called the Water Effect Ratio (WER) was introduced(USEPA, 1994). Using WER leads to the definition of *site specific standards*.

The WER has now become the recommended method for defining standards for metals. The USEPA recommends, in 40 CFR 131, that states use *dissolved metals* for the site specific standards. The term *site* is synonymous with a state's *segment*, i.e., the segments of the Des Plaines River from the confluence of the river with the Chicago Sanitary and Ship Canal to the Brandon Road Lock and Dam and from the Brandon Dam to the I-55 bridge are perfectly suitable and eligible for the site specific standard definition (USEPA, 1994). For metals and ammonium, the site specificity is inherent because the standards are related to other site specific water quality parameters of the water body (hardness for toxic metals and pH and temperature for ammonium).

Although the WER concept has been recommended by the USEPA for metals (including metalloids such as arsenic), *"this guidance is applicable to pollutants other than metals with appropriate modifications"*[3]. The magnitude of the WER can be as low as WER = 0.09 (for lead) to WER = 1.0. WER of 1.0 implies that the toxic fraction, to which the standard is to be applied, is the total metal concentration. The USEPA (1994) *Water Quality Standards Handbook* presented the magnitudes of WER as compiled by the USEPA; however, these values may not be applicable to the Des Plaines River segments being investigated.

The most preferable method is to use dissolved metal concentrations and compare them with the standard. If dissolved concentrations are not routinely measured, the site specific (statistical) WER can be calculated by the well known partitioning equation (Thomann and Mueller, 1997; and Novotny and Witte, 1997) or its simplified linear form

Table 2.1 Compilation of Numeric Illinois State Standards (Draft) and h Federal Aquatic Life Protection and Water Contact Criteria

Parameter	Illinois General Use Standards		Federal Aquatic Life Protection Criteria		Illinois Secondary Contact and Indigenous Aquatic Use Standards Title 35:Env. Protection, C:Wat.Pollution, CH1
	Title 35:Env. Protection, C:Wat.Pollution, CH. 1		40 CFR 131		
pH (units =- log $[H^+]$)	6.5 - 9		6.5 - 9		6 - 9
Phosphorus (mg/L)	0.05 (streams and shallow pools excluded)		Draft criteria are site specific		NA
Dissolved Oxygen (mg/L)	5.0 (minimum), 6.0 (for 16 hours on any day) (Permissible excursion at flows less than $Q_{7\text{-}10}$)		Early life stages: 7 day mean - 6.0 1 day minimum - 5.0 Other life 7 day minimum - 4.0 1 day minimum - 3.0		4.0 3.0 (Calumet Canal) (Permissible excursion at flows less than $Q_{7\text{-}10}$)
Toxic compounds	Acute (draft)	Chronic (draft)	Acute	Chronic	
Arsenic (µg/L) trivalent-dissolved	360*<u>1.0</u>	190*<u>1.0</u>	360	190	1000 (total)
Cadmium (dissolved)[1)] (µg/L)	exp[A+Bln(H)]x <u>{1.138672-[(lnH)(0.041838]}*</u> A=--2.918 B= 1.128	exp[A+Bln(H)]x <u>{1.101672-[(lnH)(0.041838]}*</u> A= -3.490 B= 0.7852	A= -3.828 B= 1.128	A=-3.490 B=0.7852	150 (total)
Chromium (total hexavalent) (µg/L)	16	11	16	11	300
Chromium (trivalent-dissolved)[1)] (µg/L)	exp[A+Bln(H)]x <u>0.316*</u> A= 3.688 B= 0.819	exp[A+Bln(H)]x <u>0.860*</u> A=1.561 B=0.819	A=3.688 B=0.819	A=1.561 B=0.819	1000 (total)
Copper (dissolved)[1)] (µg/L)	exp[A+Bln(H)]x <u>0.96*</u> A= -1.464 B= 0.9422	exp[A+Bln(H)]x <u>0.96*</u> A= -1.465 B= 0.8545	A= -1.464 B=0.9422	A=-1.465 B=0.8545	1000 (total)

Parameter	Illinois General Use Standards Acute	Illinois General Use Standards Chronic	Federal Acute	Federal Chronic	Illinois Secondary Contact and Indigenous Aquatic Use
Cyanide (µg/L)	22	5.2	22(Total)	5.2(Total)	100 (total)
Lead (dissolvedl)[1)] (µg/L)	exp[A+Bln(H)]x {1.46203- ln(H)(0.1457120]}* A= -1.301 B=1.273	exp[A+Bln(H)]x {1.46203- [(lnH)(0.145712)]} * A=-2.863 B=1.273	A=-1.46 B=1.273	A=-4.705 B=1.273	100 (total)
Mercury (dissolved) (µg/L)	2.6x0.85*=2.2	1.3x0.85=1.1*	2.4	0.12	0.5 (Total)
Nickel (dissolved)[1)] (µg/L)	exp[A+Bln(H)]x 0.998* A=0.5173 B=0.8460	exp[A+Bln(H)]x 0.997* A=-2.286 B=0.8460	A=3.3612 B=0.846	A=1.1645 B=0.846	1000 (total)
TRC (µg/L)	19	11			
Zinc (dissolved) (µg/L)	exp[A+Bln(H)]x 0.978* A=0.9035 B=0.8473	exp[A+Bln(H)]x 0.986* A=-0.8165 B=0.8473	A=0.8604 B=0.8473	A=0.7614 B=0.8473	1000(total)
Benzene (µg/L)	4200	860			
Ethylbenzene (µg/L)	150	14			
Toluene (µg/L)	2000	600			
Xylene (µg/L)	920	360			

Footnotes

ln[H] is a natural logarithm of hardness

*Conversion factor (translator) for dissolved metals

Conversion factor means the percent of the total recoverable metal found as dissolved metal in the toxicity tests to derive water quality standards. These values are listed as components of the dissolved metals water quality standards to convert the total metals water quality to dissolved standards and were obtained from the USEPA water quality criteria. In the federal criteria this parameter is represented by the Water Effect Ratio.

Metals translator means the fraction of total metal in the effluent or downstream water that is dissolved. The reasons for using a metals translator is to allow the calculation of total metal permit limits from a dissolved metal water quality standard. In the absence of site specific data for the effluent or receiving water body, the metals translator is the reciprocal of the conversion factor. **If dissolved metal concentrations are used, the underlined conversion factor (translator) needs to be used when dissolved concentrations are compared to the standard. The translator needs not to be used when total concentrations are compared to a standard.**

Table 2.1 - Continued

Parameter	Illinois General Use Standards	Federal Aquatic life and Human Health Protection Criteria	Illinois Secondary Contact and Indigenous Aquatic Use Standards
Barium (total) (mg/L)	5.0		5.0
Boron (total) (mg/L)	1.0		
Chloride (mg/L)	500		
Fluoride (mg/L)	1.4		15
Iron (dissolved) (mg/L)	1.0	1.0	2.0 (total) , 0.5 (dissolv.)
Manganese (total)(mg/L)	1.0		1.0
Oil, fats and grease (mg/L)			15.0
Phenols (mg/L)	0.1		0.3
Selenium (total) (mg/L)	1.0		1.0
Silver (total)[1] (μg/L)	5.0	A=-6.52 B=1.72	1100
Sulfate (mg/L)	500		
Total Dissolved Solids (mg/L)	1000		1500
Coliform[2] (No/100ml)	200 (May - October) (geometric mean) 400 (max 10 % of samples in any 30 day period) Fecal coliforms	126 (geometric mean of 5 samples over a 30 day period) E. coli - Risk based geometric mean and maximum single value (see Chapter 6)	Repealed
Temperature	32°C (Apr.-Nov.) 16°C (Dec. - March)[3]	Local and site specific	> 34°C ≤5% of time ≤ 37.8 at all times
Total ammonium as N (mg/L)	calculated[4] [5]	calculated[5]	calculated[4]
Nitrate (drinking water) mg/L as N	10	10	
Un-ionized ammonia as N (mg/L)[3]	Superceded by the adoption of the federal criteria[4] [5] for total ammonium	Superceded by the 1999 federal criteria[5] for total ammonium	0.1
Radioactivity Gross beta (pCi/l) Radium 226 (pCi/l) Strontium 90 (pCi/l)	 100 1 2		

Reference Water Bodies

Reference water bodies are selected water bodies within the ecoregion that are (1) of the same morphological and ecological character as the investigated water body, and (2) are the least impacted or unimpacted by human polluting activities and discharges. The water body assessment and monitoring activities of the UAA processes also extend to the reference water bodies.

The reference water bodies and conditions in a UAA are needed:

- To ascertain the ecologic potential of the studied impaired water body (i.e., the Des Plaines River); and/or
- To invoke Reason 1 of the UAA in a situation where natural water quality and/or its water quality parameters do not meet the nationwide or statewide chemical standards.

The water quality and biological characteristics derived from monitoring reference water bodies - reference conditions - are used for (1) estimating background and natural water conditions; (2) as a reference for bioassessment using biotic indices; and (3) as a measure of the potentially attainable water quality that the investigating stream should be approaching but not necessarily reaching. The goal of the UAA is *not* to return a water body heavily impacted by urbanization or other large scale watershed changes to natural pristine conditions. This goal would be unrealistic and unattainable. Rather the UAA should find what is the best water use, considering the irreversible changes in the watershed and physical irreversible modifications of the receiving water body.

Natural water quality and water body conditions are expressed as the physical, chemical, and biological characteristics that result from interactions within a natural ecosystem. Factors, such as land surface form, mineral availability, vegetative cover, animal and aquatic biota communities, and climate affects the natural water quality. Karr and Chu (1999) state that in multimetric biological assessment, the reference condition equates with biological integrity - defined as the condition at the site able to support and sustain a balanced, integrated, and adaptive biological system having the full range of elements and processes expected for the region. Biological integrity is the product of the ecological and evolutionary process at a site in the relative absence of human influence.

Estimating background/natural water quality is key to a UAA since, legally, use-based water quality standards may not be enforceable if the violation is due to natural causes (Reason # 1 of the UAA regulations for change of the use and/or the standards). A distinction should be made between "natural" and "background" water quality.

Natural water quality and constituent loads (note that the "pollution" and "pollutant" definitions in the Clean Water Act do not apply to natural water quality even in cases where apparent impairment is evident) vary from region to region and can be related to morphological, geographical, and ecological characteristics. Ecoregions represent relatively homogeneous geographical areas with similar structure and function between environmental characteristics (Omernik, 1987; Gallant et al., 1989). Within an ecoregion it is reasonable to expect similar natural water quality in bodies that have similar morphological characteristics and stream order.

Natural loads of constituents are topically related to the unimpacted four native land categories (Novotny, 2003): (1) Woodland, (2) Prairie, (3) Arid land (including deserts), and (4) Wetlands. The natural activities that affect the concentrations of chemical constituents in water include weathering, erosion, volcanic activity, and biological activity. Chemicals with sources in natural pathways include suspended solids and turbidity, heavy metals, dissolved oxygen, organics and nutrients. Complex organic chemicals such as PCBs, pesticides, fertilizers, may enter receiving waters through natural processes (e.g., erosion) but are initially introduced into the environment only through anthropogenic processes. Any apparent background concentrations of these chemicals cannot be considered natural and the question remains whether these sources can be controlled or not.

Natural metal concentrations or dissolved oxygen in streams may sometimes exceed the chronic or even acute toxicity standards, especially when considering extreme occurrences (once in3 years). These issues must be addressed by a UAA.

Box 2.1 Example of natural water quality and causes that may allow modification of the designated use and/or standards (Novotny et al., 1997) :

1. Naturally ephemeral streams with longer periods of no flow. The use could be modified to reflect the life forms typical for natural ephemeral water bodies, including wildlife.
2. Naturally dystrophic streams draining wetlands that have low dissolved oxygen conditions and/or could be naturally acidic.
3. Streams draining watersheds with ore deposits may have high concentrations of metals.
4. Streams in arid watersheds that carry very large natural loads of sediments.
5. Bacterial contamination caused by water fowl.

Some background loads are *legacy* loads such as atmospheric PCB deposition that is mostly global and ambiguous. Box 2.1 lists some possible types of natural water quality that could be considered as water quality impairment but not by pollution or pollutant in the sense of definitions in the Clean Water Act and should be addressed and possibly disposed by a UAA prior to embarking on a TMDL.

Karr and Chu (1999) and a number of other authors, have pointed out that there may be few, if any, places left that have not been influenced by human activities. Definition and selection of reference sites, and measuring the reference conditions may use current and/or historical data or theoretical models. Arbitrary selection of reference sites, especially if they are degraded, rather than looking over a wide area for minimally disturbed sites, and inaccurate ranking of the sites should be avoided.

The reference conditions can be obtained:

1. From monitoring of morphologically similar unimpacted or least impacted water bodies; and/or
2. From historical records of pre-development conditions; and/or
3. From monitoring upstream unimpacted water quality.

Regional Reference Sites

Box 2.2 - Regional Reference Site Selection (USEPA, 1991b)

To determine specific regional reference sites for streams, candidate watersheds are selected from the appropriate maps and evaluated to determine if they are typical for the region. An evaluation of the level of human disturbance is made and a number of relatively undisturbed reference sites are selected from the candidate sites. Generally, watersheds are chosen as regional reference sites when they fall entirely within typical areas of the region. Candidate sites are then selected by aerial and ground surveys. Identification of candidate sites is based on:

1) absence of human disturbance
2) stream size
3) type of stream channel
4) location within a natural or political refuge
5) historical records of resident biota and possible migration barriers.

Final selection of reference sites depends on determination of minimal disturbance derived from habitat evaluation made during site visits. For example, indicators of good quality streams in forested ecoregions include:

1) extensive, old natural riparian vegetation
2) relatively high heterogeneity in channel width and depth
3) abundant large woody debris, coarse bottom substrate, or overhanging vegetation
4) relatively high or constant discharge
5) relatively clear waters with natural color and odor
6) abundant diatom, insect and fish assemblages, and
7) presence of piscivorous birds and mammals.

To develop water quality criteria, the UAA should consider reference conditions. In some cases, pre-development conditions may serve as a reference, or a reference water body is selected from morphologically similar water bodies least impacted by human activities and pollution located in the same ecoregion.

Regionally attainable water quality can be approximated from physical, chemical, and biological (including bacteriological) quality of a morphologically similar water body that is minimally affected by human activities. Steps to estimate regional reference attainable water quality were outlined by Gallant et al. (1999) and listed in Novotny et al. (1997). Box 2.2 depicts the process leading to selection of regional reference sites.

Available Information on Pre-development Reference Conditions for the Des Plaines River

The Des Plaines River watershed and the investigated segment of the Lower Des Plaines River are located in the Central Cornbelt Plains ecoregion (Omernik, 1987). As stated in Chapter 1, historic annals from more than one hundred years ago described the Lower Des Plaines River at Lockport as a small stream that received its water mostly from marshes. The river had sluggish currents and since the end of the nineteenth century has been receiving sewage from the Chicago metropolitan area.

The earliest chemical analyses of the Des Plaines River water quality at Lockport were reported by Palmer (1903). The measurements included total solids(TS), suspended solids (SS) and dissolved solids (DS), total volatile solids (TVS) and volatile suspended (VSS) solids, chemical oxygen demand (COD), and nitrogen compounds. Table 2.3 presents a statistical summary of Palmer's data from 1897 to 1899.

Table 2.3 Water quality of the Des Plaines River at Lockport more than 100 years ago (Palmer, 1903)

Year	Suspended solids, mg/L	Total volatile solids, mg/L	COD[a] mg/L	Total ammonium, mg/L	Organic N mg/L	Nitrate mg/L
1897 average	11.3	37.6	11	0.46	0.92	0.84
range	0.4 - 393	12.8 - 68	6.5 - 35.7	0.2 - 1.12	0.55 - 2.83	0.1 - 3.4
1898 average	35	53.9	9.4	0.408	0.83	0.6
range	0.4 - 88.8	25.6 - 104.8	5.2 - 21.0	0.25 - 0.8	0.52 - 2.4	0.1 - 2.25
1899 average	21.6	49.2	12.9	0.48	1.0	0.36
	0.4 - 230	19.6 - 126	5.3 - 23.8	0.21 - 1.0	0.57 - 2.87	0.1 - 1.4

[a] Oxidizing agent was potassium permanganate, today's methods use chromic acid (di-chromate) as an oxidant that is more potent.

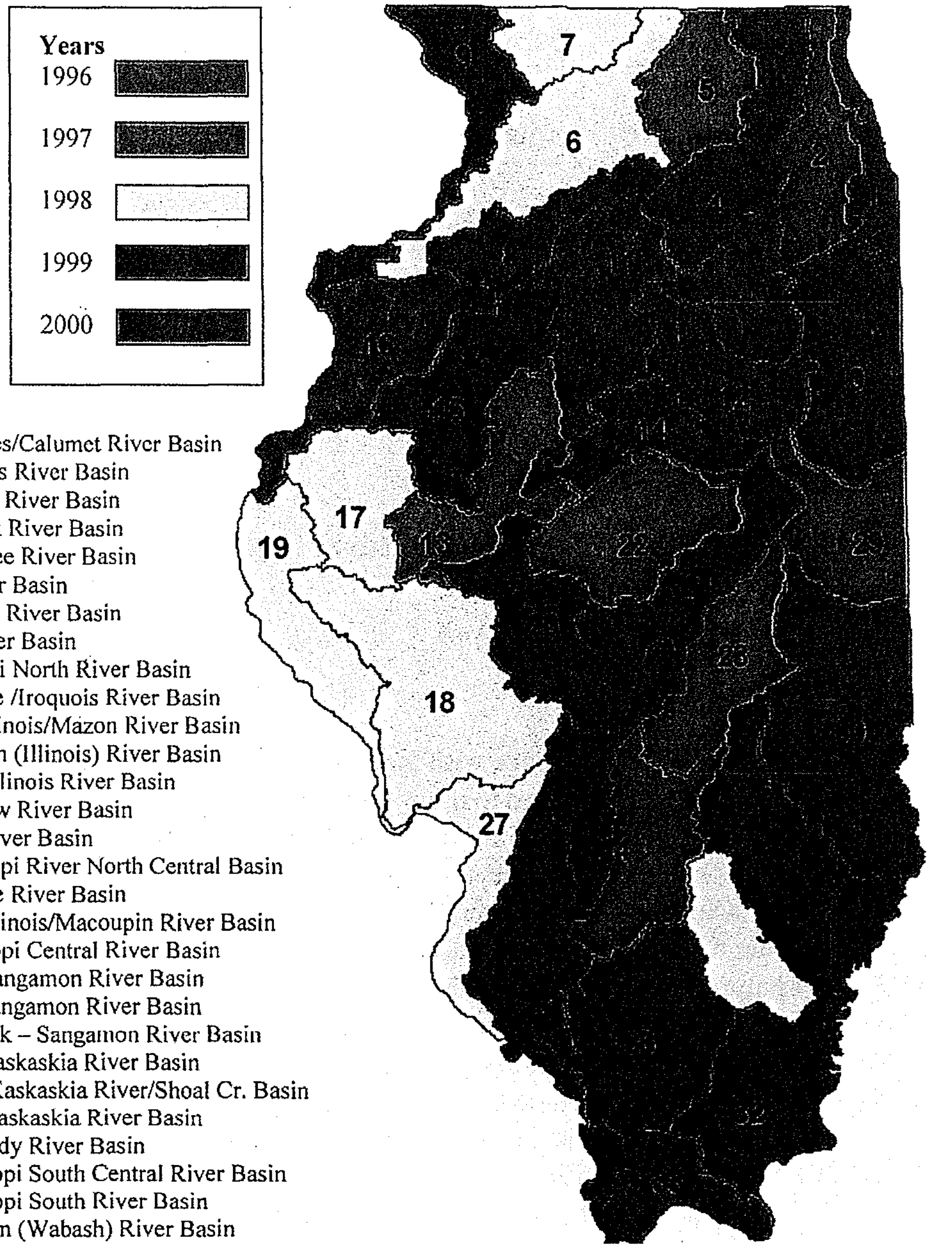
Few years after the Palmer's survey's had been conducted, the water quality of the Lower Des Plaines River was dramatically altered by the Chicago Sanitary and Ship Canal. Even Palmer's study does not reflect the pre-development conditions because the river was affected by the operation of the Illinois-Michigan canal and a portion of the river was rerouted in the late 1800s to make space for the CSSC. It can be concluded that reliable data on the pre-development water quality conditions are not available.

Reference Water Bodies in Illinois

Reference Water Bodies and Conditions. Based on the preceding discussion, the predevelopment conditions provide only an insight as to the water quality recovery limits. Unfortunately, no quantitative water quality data exists from the period prior to building the Illinois and Michigan canal. The data reported in Table 2.3 represent a situation for which some reversal of flows had occurred and Chicago raw sewage was discharged into the I-M canal and subsequently into the Des Plaines River. If the reversal of the flow by the CSSC and urban development had not occurred, the immediate watershed would have been a mixture of prairies, low land forests and wetlands and the river itself would be a sluggish wetland affected stream. To allow agricultural development, the wetlands would have to be drained. Thus, reverting the river into pre-development conditions would require an extensive wetland restoration which most likely is not possible today.

Wetland streams are typically dystrophic, i.e., they exhibit low dissolved oxygen and nutrient concentrations. They are also characterized by darker colors and higher concentrations of dissolved organics. Typically, pH is less than neutral. Thus, the key issue of the UAA is to find optimum balanced aquatic life that would sustainably propagate and do well in the Lower Des Plaines River and its major tributaries. Consequently, reverting the Des Plaines River back to its original status would not completely resolve the water quality problems. On the other hand, there is no doubt that the causes of the present dissolved oxygen and other problems in the Lower Des Plaines River are anthropogenic and means are available to maintain the dissolved oxygen in the canal and the river at levels that would not be injurious to aquatic life.

Figure 2.3. shows a map of the location of the Des Plaines River and selected reference watershed. The following reference water bodies were used: Kankakee River, Green River, Mackinaw River, Rock River, and Fox River.



1. Great Lakes/Calumet River Basin
2. Des Plaines River Basin
3. Upper Fox River Basin
4. Lower Fox River Basin
5. Kishwaukee River Basin
6. Rock River Basin
7. Pecatonica River Basin
8. Green River Basin
9. Mississippi North River Basin
10. Kankakee /Iroquois River Basin
11. Upper Illinois/Mazon River Basin
12. Vermilion (Illinois) River Basin
13. Middle Illinois River Basin
14. Mackinaw River Basin
15. Spoon River Basin
16. Mississippi River North Central Basin
17. La Moine River Basin
18. Lower Illinois/Macoupin River Basin
19. Mississippi Central River Basin
20. Lower Sangamon River Basin
21. Upper Sangamon River Basin
22. Salt Creek – Sangamon River Basin
23. Upper Kaskaskia River Basin
24. Middle Kaskaskia River/Shoal Cr. Basin
25. Lower Kaskaskia River Basin
26. Big Muddy River Basin
27. Mississippi South Central River Basin
28. Mississippi South River Basin
29. Vermilion (Wabash) River Basin

Figure 2.3 Des Plaines River and the Reference Stream/Watersheds

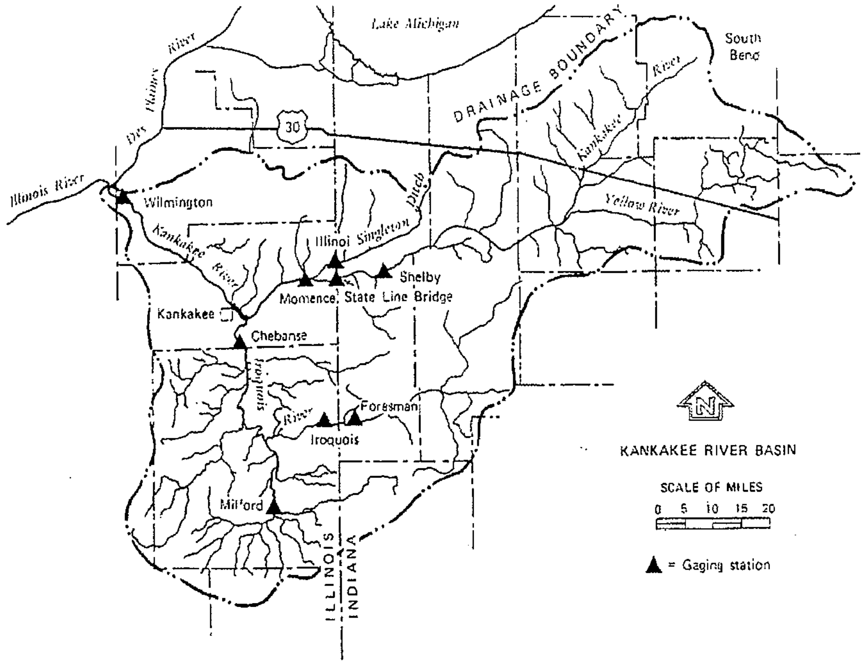


Figure 2.4 Map of the Kankakee River watershed

Kankakee River

The Kankakee River at the confluence with the Des Plaines River is the closest potential reference water body. The Kankakee River used to drain the "Grand Marsh" that encompassed approximately 400,000 acres and ranged from 3 to 5 miles in width (Ivens, et al., 1981). The nature of the marsh caused the river to change the course continuously. Most of the pre-settlement watershed was a prairie. Today, the Kankakee River watershed in Indiana is drained and converted into agricultural land. In Illinois, the river has been used as a scenic, cultural and recreational resource and in some reaches left in a natural state. The river in the Kankakee County, upstream of the confluence with the Des Plaines River, is noted for high water quality and biologists rank most of the Kankakee River along with some of its tributaries as "highly valued natural resources." (Illinois Department of Natural Resources, 2001). The river has now more siltation due to agricultural practices in Indiana. However, not all of the sediment in the Kankakee River comes from Indiana; a significant part of the sediment load originates from sources in Illinois (Ivens et al., 1981) primarily from the Iroquois River. Thus, the best reference condition is the reach between the state line and the confluence with the Iroquois River. The watershed area of the Kankakee River is 5,165 sq mi, from which 42% is in Illinois and 58% in Indiana. The river has a total length of about 150 miles, with 59 miles in Illinois.

Nearly 88% of the sampled stream miles in the Kankakee drainage "fully support " the Illinois General Use as determined by the Illinois EPA and 231,005 acres of the watershed have been designated a resource rich area. The land use distribution in the Illinois part of the watershed is as follows:

Cropland	77.6%
Grassland	15.8%
Urban/built-up	2.5%
Bottomland forest	0.8%
Nonforested wetlands	0.5%
Water	0.5%

The geologic materials of the Kankakee River basin consist of glacial deposits overlying Paleozoic bedrock. In Illinois, most of the bedrock is Silurian age dolomite, and in Indiana much of the bedrock is Devonian age shale. The most important geologic event shaping the landscape and the character of the deposits in the basin was the ancient "Kankakee Flood.," that occurred during glacial melting about 16,000 to 13,000 years ago. During this period, the retreating glacial lobes constructed numerous moraines, including the Valparaiso moraines located along the northern portion of the Kankakee River. The flood deposited thick sand in a wide belt along the Kankakee River resulting in sandy sediments extending from the City of Kankakee to South Bend, Indiana. This extensive sandy deposit is the primary source of sediments now residing in the Kankakee River.

For this UAA, the water quality monitoring site located at Momence was available as a reference site. The site is located in a relatively scenic and recreational area. The reach between the state line and Momence is a naturally meandering stream with a sandy bottom, traversing an area of timber and relatively undisturbed wetlands, known as the "Momence Wetlands." However, in view of large scale modification and wetland drainage for agriculture upstream in Indiana, the Kankakee River at Momence cannot be considered as an "undisturbed/unimpacted" stream. More or less, it may be a stream the Des Plaines River might look like if urbanization and flow reversal from the Chicago River had not occurred. Thus, this site is used in this study to document, as close as possible, the chemical and bacteriological integrity reference conditions of a stream least impacted by urbanization but is not considered as a goal for the Lower des Plaines River that is heavily impacted by navigation.

Mackinaw River

The Mackinaw River originates in Ford county near Sibley and winds approximately 130 miles in a westerly direction before joining the Illinois River near Pekin. The basin area is approximately 1,138 sq miles. The land use distribution in the watershed is as follows (Illinois Department of Natural Resources, 2001) :

Cropland	76.9%
Grassland	13.5%
Upland forest	4.9%
Urban Built-up	2.3%
Water`	1.3%
Bottomland forest	1.8%
Nonforested wetland	0.3%

The Mackinaw River is considered one of the best examples of a prairie stream left in Illinois and 136.4 miles have been designated as biologically significant (Figure 2.4). The macroinvertebrates found in a survey appear to be more diverse than those of many other watersheds in Illinois, which is an indication of good water quality.

However, water pollution from build-up and agricultural lands has lead to a decline in the aquatic life of the Mackinaw River, particularly mussels and fishes. Compared to other major tributaries of the Illinois River, the Mackinaw River basin has one of the highest sediment yield rates in the Illinois River basin. An estimated 2.1 million tons of sediment are delivered annually to the Illinois River (Illinois Department of Natural Resources, 1997).

In 1992, the Nature Conservancy, Illinois Department of Natural Resources, and IEPA approved the Mackinaw River Partnership which in 1996 became an official Ecosystem Partnership. The partnership receives funding from the IDNR through the Clean Water Act Section 319 programs.

Figure 2.5 Mackinaw River

The Mackinaw River is considered as a reference stream in Illinois. Its relatively good water quality and ongoing preservation/restoration programs make the river an example of attainable integrity of a small to medium stream (Figure 2.5). However, its much smaller size than the Des Plaines River precludes its use for chemical assessment. The data is used in this study as a reference for bacteriological contamination.

Green River

The drainage basin of the Green River covers 1131 sq mi. The soils consist of a lake plain of sand and gravel outwash from the Wisconsin glacier. The river course follows the northern boundary line of the Wisconsin terminal moraine in a general southwesterly direction. The headwaters originate north of Compton in the southeastern corner of Lee County and the stream enters the Rock River approximately two miles west of Green Rock. Before draining activities in the late 1880s, the river flowed through two large swamps. Except for two sections, totaling 27 miles, the river has been dredged, straightened, and reduced to a canal like environment. The latest (2002) 305(b) report rated 56 miles of the river as fully supporting (good) and 26 miles as partial support (fair).

The average width of the river is about 90 ft and the river is relatively shallow. The water is generally clear with a substrate of gravel in the undredged sections and a substrate of almost pure sand in the dredged sections. The river pollution has been gradual and not visible but silt, agricultural chemical runoff, animal, domestic, and industrial waste sources are present. The nutrient pollution has caused extensive phytoplankton blooms (Illinois DNR, 2001).

Because of the absence of municipal pollution this site was used as a reference for bacterial pollution, representing an agricultural stream.

Reference Water Bodies to Assess Impact of Navigation

One question that can be addressed at the beginning of the UAA is the role of navigation and its possible removal. Reason 4 of the UAA regulations that allows modification of the standards states:

Dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use.

Therefore, there are two issues to be addressed: (1) Possible restoration of the river to its original condition; and (2) Operating the system so that the aquatic life and primary recreation uses could be attained.

Section 303(c)(2) of the Clean Water Act provides clear guidance on the possible reversibility of the present conditions of the system and change of the designated use. This section states that when revising and/or developing new water quality standards[1]: *...Such standards shall be established taking into consideration their use[2] and value for water supplies, propagation of fish and wildlife, recreational purposes, and also taking into consideration their use and value for navigation.* Thus one may conclude that, based on the CWA:

1. Viable and economically important navigation by the CSSC appears to be a protected use. The CSSC and the Lower Des Plaines River are heavily used for navigation. Removing navigation would create a widespread economic burden and would disrupt the Chicago and Great Lakes

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[1]A "standard", according to the definition in the Clean Water Act (Section 305(c)(2)) consist of the designated use and the water quality criteria to protect the use.

[2]The context of this statement implies use of the water body not use of the standards.

commerce. Even without considering Section 303(c)(2) this would most likely trigger Reason 6 of the UAA, i.e., removing navigation could create a wide spread adverse socio-economic impact. *The AquaNova International and Hey Associates team has concluded that removing navigation from the Des Plaines River cannot be considered as a viable remedy for the water quality problems of the Des Plaines River.* The same is not true for the Illinois-Michigan canal that has been mostly abandoned and has no economic value for navigation that ceased in 1933. The legal status of this water body is uncertain and irrelevant in the context of this UAA.

2. The CSSC and the Lower Des Plaines River are used for waste conveyance in order to prevent contamination of the potable water intakes located in Lake Michigan that provide water for the Chicago metropolitan area. Although waste conveyance in the context of UAA is not considered a beneficial use, reversing the flows and creating the CSSC was the primary reason why the waterway was proposed and created in the late 1800s and early 1900s. Thus the safety of the water supply for the entire Chicago metropolitan area must be taken into consideration. However, flow reversal and wastewater conveyance impairs water supply on the Illinois River.

Thus, the century old and well functioning and managed system of the Chicago Sanitary and Ship Canal with its tributary, the Calumet Sag Canal, must be considered for the foreseeable future as an irreversible reality. Consequently, finding the way to operate the system in a way that would allow the attainment of aquatic life and recreation uses will be the task of this UAA.

However, considering navigation as an unremovable physical attribute of the Des Plaines River only allows consideration of the UAA habitat issues and some water quality modifications. It does not give relief, in the TMDL process, to discharges of pollutants or pollution into the water body and the navigational physical attributes alone may not provide a possibility to downgrade the primary recreational use and associated bacteriological standards (see Chapter 6 and the US EPA [2000, 2002] draft documents for establishing criterion for bacteria).

Reference Impounded Water Bodies

In the long run, it is not possible to remove navigation in impounded pools of the Illinois Waterway and restore the river to a natural state. Hence, the ecologic potential of the Des Plaines River cannot be directly related to a pristine unimpacted reference water body (that may not even be available near the Des Plaines River) but to some other mixed impounded water bodies[3] that are minimally impacted by pollutants. The Indices of Biotic Integrity established for these reference impounded water bodies, after a critical evaluation, will then serve as a measure of the ecologic potential of the navigational impoundments.

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[3] Well mixed unstratified impoundments are generally lakes behind the low head dams. Their ecology and water quality is different from deep stratified impoundments.

Rock River

The Rock River originates in Horicon Marsh in Dodge County, Wisconsin, and flows in a southerly direction until it enters Illinois south of Beloit. It continues to flow south and southwest across the northwestern part of Illinois, and joins the Mississippi River at Rock Island.

The total drainage area of the entire Rock River is about 10,900 sq miles of which about 6,400 sq mil is located in Illinois. The Wisconsin portion has population of about 754,000. Major population centers include Madison, Janesville, Beloit and the expansion area. Major urban centers in Illinois are Rockford (pop. 139,943), Moline (pop. 43,127), Rock Island (40,630), Sterling (15,152) and Dixon(15,134). Despite its urban centers, the Rock River basin remains largely rural in character, both in Wisconsin and Illinois. The total stream length, including the mainstem and tributaries, is 2325 miles.

Significant tributaries include the Kishwaukee River, Sugar-Pecatonica River Basins, and the Green River. The mainstem length in Illinois is 163 miles. Of the total river miles, 69 stream miles have "good" quality and 97.9 miles have fair quality. Nutrients, phosphorus in particular, suspended solids and channel modifications are the major cause of water quality problems due to agricultural runoff and flow modifications and regulations. The river is impounded, both in Wisconsin and Illinois.

Fox River

The Fox River originates in Wisconsin in Waukesha County and flows generally in a southerly direction until it joins the Upper Illinois River. The watershed is directly to the west of the Des Plaines River watershed. The river is of interest as a reference stream because of the extensive study conducted sponsored by the Illinois Department of Natural resources and USEPA on the effect of impoundments on the biotic integrity and fish assemblages (Santucci and Gephard, 2003). There are 15 dams on the Fox River, however, navigation is mostly recreational and is not quite comparable to the Lower Des Plaines River. The river and its tributaries are known to support a high diversity of aquatic organisms including 32 species of mussels and 96 species of fish.

The main stem of the Fox River in Illinois is about 115 miles. The watershed encompasses McHenry, Lake, Kane, DuPage, DeKalb, Kendall, and LaSalle counties. The upper part of the watershed is agricultural and the middle part is rapidly urbanizing due to rapid expansion of the Chicago suburbs. The largest cities in the watershed are Aurora (100,000) and Elgin. The most current assessment in the 2002 305(b) Illinois report rated 33 miles of the Fox River as full use (good) and 67 miles as partial support (fair). The primary causes of less than full use included nutrients siltation, low dissolved oxygen, flow alteration, habitat alteration, suspended solids, fecal coliforms and pH. These problems were attributed to agriculture, urban runoff, CSOs, hydrologic modifications/flow regulations, stream bank stabilization/modification and contaminated sediments. It should be pointed out that the Fox River has been classified as General Use water body.

Methodology for Water Body Assessment

Data from several agencies were used to conduct a probabilistic analysis of parameters covered by the Illinois General Use Standards found in Tables 2.1 and 2.2. The analysis was conducted using the statistical software package StatGraphics. Data from the Des Plaines River obtained from monitoring/sampling programs of the Illinois Environmental Protection Agency (IEPA) as part of the Ambient Water Quality Monitoring Network (AWQMN), the United States Geological Survey (USGS) and the Metropolitan Water Reclamation District (MWRDGC) was input into StatGraphics. A list of sampling points is included in Table 2.4 and the locations are shown in Figure 2..6.

Table 2.4 Sampling Points Used in Statistical Analysis

Code	Water Body	Agency	Location
91	Des Plaines River upstream of Lockport	MWRDGC[1)]	Material Service Access Road near Lockport Power House
92	Sanitary & Ship Canal	MWRDGC[1)]	Lockport Power House Forebay
93	Des Plaines River - Brandon Pool	MWRDGC[1)]	Joliet, Jefferson Street Bridge, Joliet
94	Des Plaines River, Dresden Pool	MWRDGC[1)]	Empress Casino Dock
95	Des Plaines River, Dresden Pool	MWRDGC[1)]	Interstate 55 Bridges
G-11	Des Plaines River, upstream from Lockport Dam	IEPA[2)]	Division St. Bridge at Lockport near Lockport Power House
GI-02	Sanitary & Ship Canal	IEPA[2)]	Lockport Power House Forebay
G-23	Des Plaines River, Brandon Pool	IEPA[2)]	Ruby Street Bridge, Route 53 in Joliet
G-39	Des Plaines River, upstream of Lockport	IEPA AWQMN	Barry Point Road, Riverside
F-02	Kankakee River	IEPA AWQMN	Route 17 Bridge, Momence

1) MWRDGC stations 91=95 are sampled weekly

2) IEPA stations are sampled nine times per year

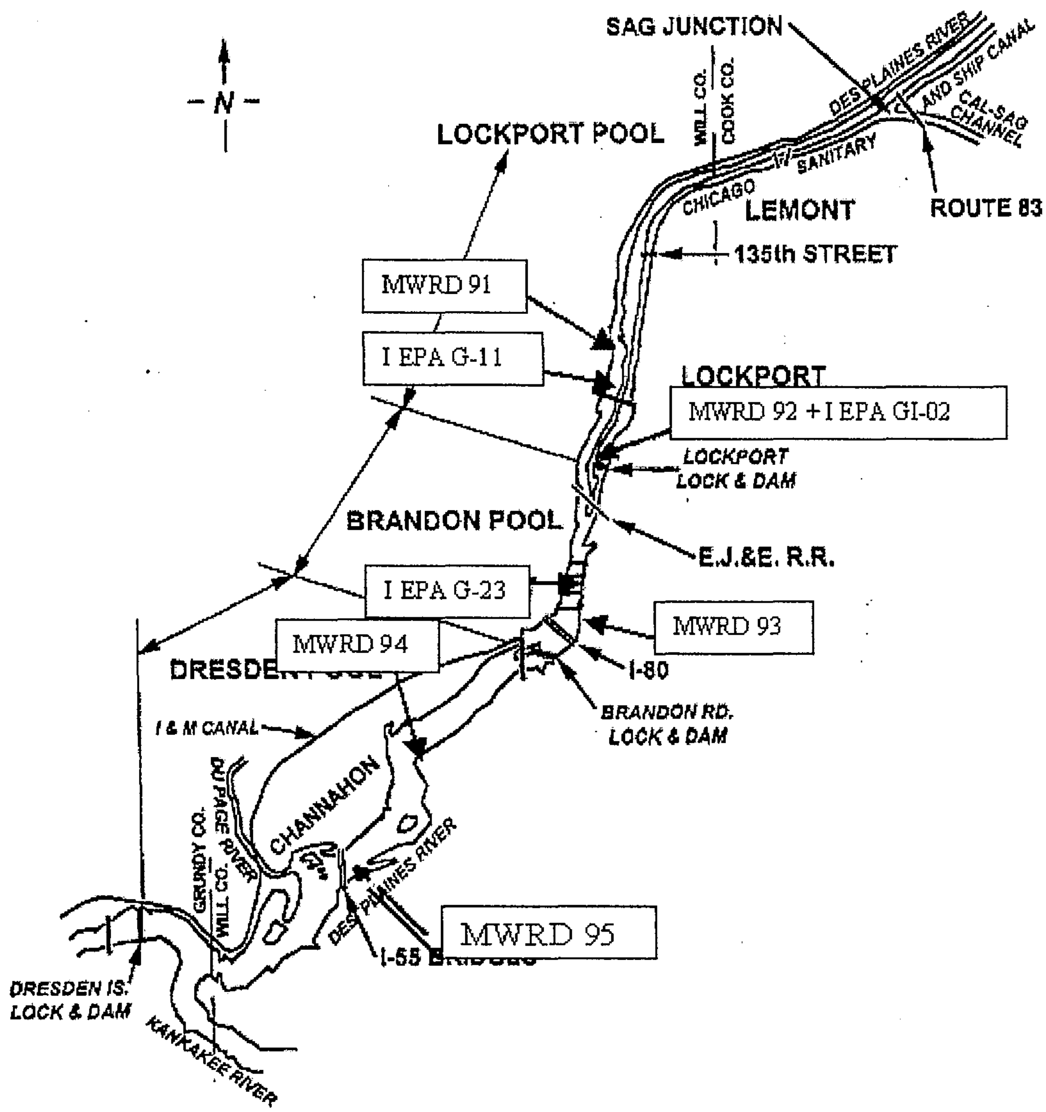

Figure 2.6 Location of Sampling Sites in the Des Plaines River

The key sampling points based on which the use attainability analysis has been evaluated are those located in the segments of the Des Plaines River between the Lockport Dam and the I-55 Bridge. The reference site on the Kankakee River defines the reference conditions for this preliminary analysis. Analysis of data in the river upstream of Lockport and in the CSSC is for comparative purposes.

The report evaluates the water quality data obtained from the agencies for compliance with the Illinois General Use Standards. If a parameter complies with the General Use it can be implicitly assumed that it also complies with the Secondary Contact and Indigenous Aquatic Life Use for which the standards are less stringent. Some parameters (e.g., bacteria) have only a General Use standard.

Statistical probability plots of both IEPA and MWRDGC data for the last five years, i.e., 1995 - 2000 were produced for each parameter and included in Appendix B. The period of record varied for each parameter, but a guideline of a five-year record limitation (1995 - 2000) recommended by the subcommittee of experts for this project, was used for all Des Plaines River sites. In the case of the reference sites, all existing data were used in the statistical analysis. This is due to the fact that tha changes in most reference watersheds are not rapid (they should be least impacted by human actions) and the data base might be insufficient if restricted only to the last five years.

Some MWRDGC stations had less than five years of data; however, because of the higher frequency of data acquisition there were enough data points for the analysis. In most cases, the log value of the parameter was used because the logarithmic transformation of the water quality data followed a log-normal distribution. This is exhibited on the plot by data being arranged in an approximate straight line. Temperature and pH did not follow a log-normal distribution. pH, being already a logarithm of the reciprocal of the hydrogen ion concentration, was fitted to a normal distribution. Normal distribution defined from $-\infty$ to $+\infty$ does not fit well with parameters that have a near physical limit such as temperature. Log normal distribution is defined from 0 to $+\infty$.

Percentiles for Comparison with Standards

As stated previously, it is not possible to consider standards as never to be exceeded although if no data exceeded the standard it would be, obviously, a good but not unbiased indication of compliance. However, the three dimensional nature of the standard and its application must be considered for priority pollutants. Note the probability of not being exceeded $X = p(C<C(max))$ equals $1 - p(C \geq C(max))$. If one exceedance is allowed by the criteria regulations, this also implies that one or two values that equal the standard are also allowed. Therefore, the probability of required compliance was set at 99.8 percent of measured values of being less than the standard. For dissolved oxygen the allowable exceedance is reversed, i.e., the limit is C(min). In a practical sense, the probability of exceedance, 1 - X, is the frequency times duration. Since duration is assumed generally as one day (one grab sample) then the probability of the nonexceedence is 1 - probability of (exceedence + equality) = 1 - 0.2 = 99.8 % for toxic priority pollutants, that also includes Criterion

Continuous Concentration (CCC) limit for ammonium and the probability of allowable excursion for the "absolute minimum" of dissolved oxygen[4].

For nonpriority pollutants the allowable exceedance has not been specified. The guidelines for the CWA Section 305(b) reports allow 10 % of data excursions for classification of water bodies as being in compliance. No other permissible frequencies have been included in the federal criteria regulation for nontoxic pollutants. As shown on Figure 2.2, the difference between the 90 and 99.8 percentile concentrations may be as much as one order of magnitude if the concentrations follow a log-normal probability distribution. Using 10 % allowable excursions underestimates the degree of impairment and will not be used for estimating exceedences of toxic priority pollutants and dissolved oxygen. For nontoxic pollutants, a scientific judgement on the compliance will be used if the probability of exceedance is more than 0.2 % but less than 10 %.

Tier I - Screening Analysis

Calculation of Site Specific Standards

Metals. The standards are related to and calculated from hardness. Hardness is a log-normally distributed parameter characterized by the geometric mean and log standard deviation. Consequently, the standard is also a statistical variable. Nevertheless, research done at Marquette University used statistical and Monte Carlo methodologies and found that the probability of a standard exceedance can be reliably ascertained using the (geometric) average of hardness(Bartosova and Novotny, 2000).

Table 2.5 presents the metal criteria for the Des Plaines River sites calculated from average hardness. The standards listed in Table 2.5 are for dissolved metals. When dissolved metals are compared with the standards, the Illinois standards have to be multiplied by the conversion factor specified in Table 2.1 for the Illinois General Use (Table 2.5).

Total Ammonium. The criteria for ammonium are, as the previous standards were, related to pH for CMC values and pH and temperature for CCC values (see Table 2.1). The criteria for the Des Plaines River were calculated for salmonid fish absent and early life present conditions. The ammonium concentrations in the river during high temperature conditions (summer) are lower due to the enhanced nitrification in the treatment plants and in the river itself. However, temperatures above 22°C may suppress nitrification (Zanoni, 1968). Higher concentrations of ammonium are typically found during cold winter conditions. This will be considered when judgement on the attainability is made.

For the evaluation this study used the federal USEPA water quality criteria because the new Illinois water quality standard for ammonium was not issued until Novemeber 2002, long after the report analysis was conducted. The new Illinois standard is similar if not identical to the federal criterion.

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[4] The use of the same probability of allowable excursions for dissolved oxygen and priority pollutants is based on the facts that (a) oxygen depletion is toxic, and (2) the allowable excursions specified at the minimum low flow with a recurrence interval of once in 10 years has approximately the same probability as the frequency (probability) of allowable excursions of once in 3 years.

Probability Plots

An example of an individual probability plot for a toxic compound is shown in Figure 2.7. The chronic CCC standard is shown as the lower concentration represented by a dashed line and the acute standard as the higher acute CMC value shown as the solid bold line. This methodology was followed for all parameters. Likewise, the plots for dissolved oxygen were altered to show both the minimum 5.0 mg/L standard and the 6.0 mg/L sixteen hour standard, as seen in Figure 2.8. The decision on excursions from the standards is made from visual fitting.

Using a line of the best fit estimated by the StatGraphic software is not feasible because the water quality evaluation is focusing solely on the extreme values while the line of the best fit calculated by the software considers all values that were included in the plot, including outliers. Therefore, professional judgment is superior to a calculated extreme value. This is documented on the figure by the thin (all points considered) and bold (best fit) lines.

Table 2,5 Acute and Chronic Toxicity Illinois Standards Derived from Average Hardness for Dissolved Metal Concentrations

	Average Hardness	Cadmium (ug/L)		Chromium (ug/L)		Copper (ug/L)	
Site	(mg $CaCO_3$/L)	Acute	Chronic	Acute	Chronic	Acute	Chronic
Reference (Kankakee)	293.50	29.59	2.28	1325.37	429.94	46.93	28.48
IEPA - G-11	284.50	28.60	2.23	1291.99	419.11	45.57	27.74
IEPA - GI-02	230.94	22.81	1.91	1089.11	353.30	37.44	23.21
IEPA - G-23	238.50	23.62	1.96	1118.22	362.74	38.60	23.86
MWRDGC 91	300.80	30.36	2.32	1352.31	438.67	48.03	29.09
MWRDGC 92	232.80	23.01	1.92	1096.29	355.62	37.73	23.37
MWRDGC 93	247.60	24.60	2.01	1153.05	374.04	39.98	24.63
MWRDGC 94	250.40	24.90	2.03	1163.72	377.50	40.41	24.87
MWRDGC 95	246.40	24.47	2.01	1148.47	372.55	39.80	24.53
USGS Riverside	267.20	26.72	2.13	1227.28	398.12	42.96	26.29

	Average Hardness	Lead (ug/L)		Nickel (ug/L)		Zinc (ug/L)	
Site	(mg $CaCO_3$/L)	Acute	Chronic	Acute	Chronic	Acute	Chronic
Reference (Kankakee)	293.50	239.0	50.4	204.8	12.4	304.25	54.48
IEPA - G-11	284.50	231.4	48.8	199.5	12.1	296.30	53.05
IEPA - GI-02	230.94	185.9	39.2	167.2	10.1	248.31	44.47
IEPA - G-23	238.50	192.3	40.5	171.8	10.4	255.17	45.70
MWRDGC 91	300.80	245.2	51.7	209.1	12.7	310.60	55.63
MWRDGC 92	232.80	187.4	39.5	168.4	10.2	249.95	44.76
MWRDGC 93	247.60	200.0	42.2	177.4	10.7	263.39	47.17
MWRDGC 94	250.40	202.4	42.7	179.1	10.8	265.92	47.62
MWRDGC 95	246.40	199.0	41.9	176.7	10.7	262.32	46.98
USGS Riverside	267.20	216.7	45.7	189.2	11.5	280.96	50.32

Probabilistic Analysis

Probability plots for all selected sites are grouped by parameters in Appendix. B. The data sets in some cases were incomplete or insufficient to provide a probabilistic analysis (as was the case for parameters in which many of the data points were at or below the detection limit). In either case, the record of the sampling site is given with a brief explanation of the data set. Probability plots were not done where all data were below the detection limit.

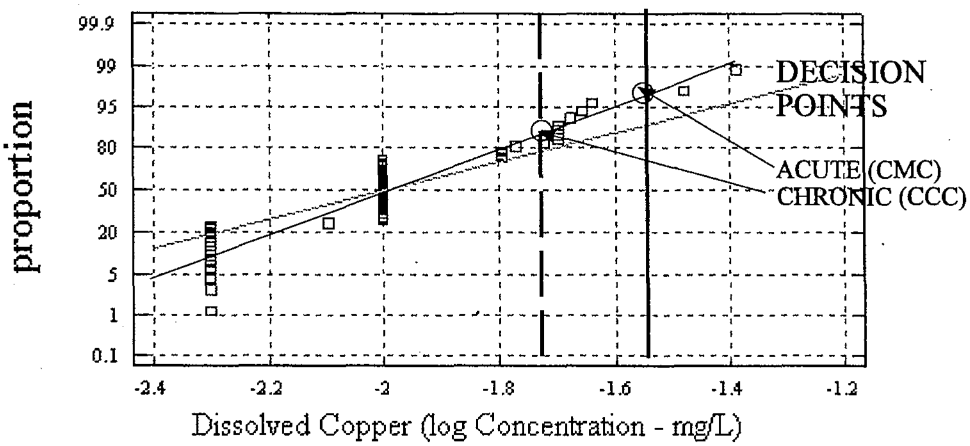

Figure 2.7 **Example of Probability Plot for Copper at MWRDGC 94 Including the Illinois General Use Acute and Chronic Toxicity Values Corresponding to the Average Hardness**

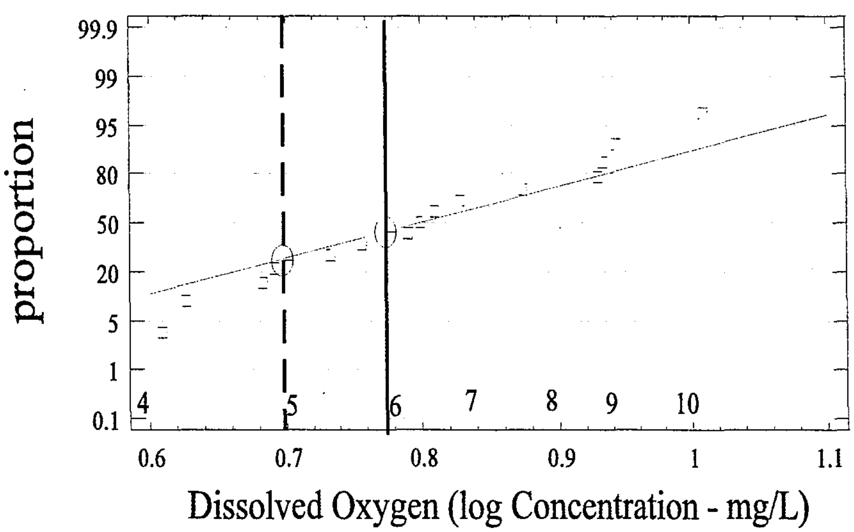

Figure 2.8 **Example of Probability Plot for Dissolved Oxygen at G-23 (Joliet) Including the Illinois General Use Standards**

Toxic compounds included in the analysis are compared to both the acute and chronic Illinois General Use Standards. Standards for metals are hardness dependent. The equations for derivation of these standards are included in Table 2.1. The standard for each individual site and dissolved metal including the average hardness, is included in Table 2.5.

The total ammonium standard was developed by the formulae taken from the recent updated federal criteria documents (USEPA, 1999). The acute and chronic criteria for ammonium are site specific because they are calculated from pH (acute) and pH and temperature (chronic).

For other toxic priority parameters, as well as other parameters, the Illinois General Use Standards are used directly in the analysis. As stated previously, acute toxicity standards are compared to a 99.8% probability of occurrence, while chronic toxicity standards are compared with the 99.4% probability of compliance. Probability plots constructed in StatGraphics are limited to the range of the data set. In some cases, standards are not shown on the probability plot due to the location of the range of values for that data set.

A summary of the parameters that meet the standards according to the probability plots for the site is included in Table 2.6. The parameters that do not meet the Illinois General Use Standards are included in Table 2.7. All parameters that meet the standards are at the 99.8 % level of the probability of not being exceeded. This means that possible exceedences could occur with a recurrence interval of more than 3 years.

Two tier evaluation of copper. In the Lower des Plaines River the IEPA measured both dissolved and total concentrations at sampling points G-11 (Lockport) and G-23 (Joliet). In addition, total and dissolved metals concentrations were available from sampling at Riverside (IEPA G-39), upstream Des Plaines River (IEPA G 02), and Kankakee (IEPA F02). The dissolved concentrations at the two sampling points in the Lower Des Plaines River have passed the 99.8 percentile probability test. However, the data on copper measured by the MWRDGC at sampling points 92, 93, 94 and 95 included only total concentrations and did not pass the 99.8 percentile test. The WER in the Illinois draft General Use Standards for copper is only 0.96; therefore, no change to the conclusion was made on the Tier 1 evaluation and copper was added to the compounds that will require further analysis. In this case the WER will be calculated from the IEPA data and used to convert the MWRGC total concentrations to their dissolved fractions.

Two tier evaluation of ammonium. Total ammonium concentrations are clearly in compliance with the Illinois and federal CMC (acute) standards and criteria. However, the evaluation of compliance with chronic (CCC) federal criteria is complicated by the fact that the time series of 30 day or 4 day average concentrations are not available. In the next section on the Tier II evaluation two methodologies will provide a more scientific and accurate assessment:

(1) Joint probability of temperature, pH and total ammonium concentrations.
(2) Monte Carlo calculation

The highest concentration of total ammonium was measured in the winter of 2000 at G-23 (Joliet) as 6 mg/L. Typical high summer ammonium concentrations are less than 1.2 mg/L. Because the CCC evaluation requires 4 or 30 days averaging of daily data that is not available, Monte Carlo simulation and CCC evaluation will be performed in the subsequent Tier II evaluation.

Tier I Evaluation and Recommendation

Parameters in Compliance

Parameters listed in Table 2.6 are meeting the Illinois General Use Standards and the federal aquatic life protection and propagation criteria. By default they also meet the current Secondary Contact and Indigenous Aquatic Life use. These water quality parameters have passed the 99.8 probability percentile test for nonexceedance in spite of the fact that some are not priority pollutants. Chloride is not a priority pollutant, organisms can tolerate extended period of higher salinity; therefore, the 97% compliance was deemed to be satisfactory (nte that the guidelines for the 305(b) reporting characterize 90% compliance for non priority pollutants as "good").

For the parameters listed in Table 2.6 the general use of the water body (aquatic life protection) has been met. The Illinois EPA should reevaluate inclusion of the metals listed in Table 2.6 and ammonium in the 303(d) list.

pH. The limits for pH for the General Use Standards (and federal criteria) are 6.5 to 9. Few exceedences of these limits were detected at MWRDGC sites 94 and 95 (Dresden Island Pool). The compliance probabilities are:

	Lower limit 6.5	Upper limit 9.0	
Reference site	99%	>99.8%	Kankakee River
IEPA GI-02	>99.8%	>99.8%	Upstream site
MWRDGC 92	>99.8%	>99.8%	Upstream site
IEPA G-23	>99.8%	>99.8%	Brandon Dam Pool
MRWD 93	>99.8%	99.8%	Brandon Dam Pool
MWRDGC 94	96%	99%	Dresden Island Pool
MWRDGC 95	96%	99%	Dresden Island Pool

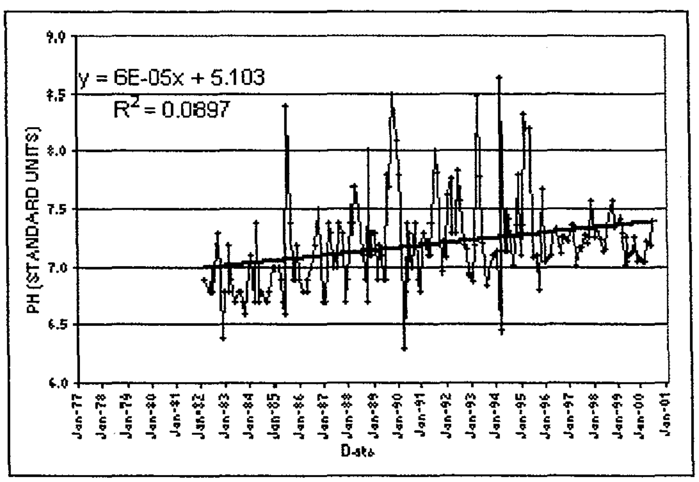

Figure 2.9 Trend of pH at IEPA G-23 in Joliet.

pH is not a priority pollutant, hence, the 99.8% rule is not applicable. Bell (1971), in a discussion of the effect of pH included in the USEPA (1986) criteria document, reported 30 day lethal value (after exposure of the organisms for 30 days) of low pH between 2.45 to 5.38 for macroinvertebrates. The criteria themselves specified that pH as low as 5.0 is unlikely to be harmful to any species unless either the concentration of free CO_2 is greater than 20 mg/L, or the water contains iron salts which are precipitated as ferric hydroxide. None of the two are likely.

We have also investigated the trend in pH at the IEPA site G-23 in downtown Joliet (Figure 2.9). The trend is increasing, meaning that the likelihood of low pH is decreasing. Thus the percent compliance with the pH standard specified above is satisfactory.

Table 2.6 Parameters Meeting Illinois General Use Standards and Federal Criteria

Parameter	Representative Sites Meeting General Use Standards	Approximate Probabilty of Compliance with General Use Standard
Arsenic	All in the Lower Des Plaines R.	>99.8%
Barium	All	>99.8%
Boron	All	>99.8%
Cadmium	All	>99.8% (CCC) [1)]
Chloride	All	97% (MWRDGC 94, 95)
Chromium (trivalent)	All	>99.8%
Cyanide (WAD CN)	MWRDGC 93, 94, 95	> 99.8 %
Fluoride	All	>99.8%
Iron	All	>99.8%
Lead	All	>99.8%
Manganese	All	>99.8%
Nickel	All	>99.8%
Phenols	MWRDGC, IEPA sites	>99.8%
Selenium	All	>99.8%
Silver	All	>99.8%
Sulfate	All	>99.8%
Tot. Ammonium as N(CMC)	All	>99.8%
Tot. Ammonium as N (CCC)	All	[2)]
Zinc	All	MWRDGCGC and IEPA sites >99.8% for total and dissolved zinc acute (CMC) standard only

[1)] Chronic standard for cadmium is 10 to 25 % below the detection limit. All measured dissolved cadmium concentrations in the last five years were at or below the detection limit, consequently it is not possible to calculate WER. Compliance with the chronic standard is impossible to ascertain but is assumed.

[2)] An exact estimation of compliance involves statistical fitting and joint probability consideration of 3 parameters Total NH_4^+, temperature and pH calculated as 30 day (4 day) averages. Furthermore, all three parameters are not pure random variables but exhibit a cyclic pattern. A scientific judgement was used in the Tier 1 analysis.

Temperature. Grab temperature data at the I-EPA, MWRDGC and Midwest Generation sites (I-55 bridge) and continuous temperature monitoring by Midwest Generation at the I-55 bridge did not reveal actual measured excursions. Furthermore, the normal or log-normal distribution do not properly represent the probability distribution of the temperature measurements. The log-normal plots have a distinct upswing tail that indicates a near physical limit (i.e., the temperature cannot physically increase under present conditions over a certain value, e.g. 40°C). The plot indicates that a temperature limit of 32°C (Illinois General Use) at GI-02, G-23, MWRDGC 92 to 95 would be met with a probability of compliance better than 99 percent. However, the MWRDGC sites in the Brandon and Dresden pools do not include data prior 2000 and IEPA does not measure temperatures in the Dresden Island pool

The Interstate - 55 bridge (mile 277.9), the end of the investigated reach, is approximately 7 miles from the cooling water outlets of the two large Joliet power plants. There is only one location in this stretch where temperature is measured occasionally during collection of grab samples at the MWRDEGC 94 site (Empress Casino). The problem of cooling water discharge on this 7 miles stretch and attainability of the general use temperature standards in the stretch of the Dresden Island pool upstream of the I-55 bridge will be addressed in the subsequent section.

Parameters That Do Not Meet the Illinois General Use Standards and Federal Aquatic Use and Contact Recreation Criteria

Several analyzed parameters did not meet the Illinois Water Quality General Use Standards and will be analyzed in more detail in Tier II - The Detailed Compliance Analysis and Simplified TMDL. Table 2.7 presents these parameters.

As proposed in the methodology, if dissolved concentrations are not measured, the total concentrations were evaluated in the Tier I analysis. If this analysis failed to find compliance and the noncompliance was marginal, WER would be estimated in the Tier II analysis and compliance will be evaluated with estimated dissolved concentrations.

Copper. Total copper concentrations at MWRDGC 92, 93, 94 and 95 did not meet the Illinois General Use Standards. The level of compliance probability were

	Acute (CMC)	Chronic (CCC)
MWRDGC 92	99 %	95 %
MWRDGC 93	> 99.8 %	99.2 %
MWRDGC 94	95 %	85%
MWRDGC 95	> 99.8 %	99 %

In general, the noncompliance is only marginal (note that the probability of noncompliance in percent is 100 - probability of compliance). Furthermore IEPA sites located at about the same location (IEPA GI-02 = MWRDGC 92 and IEPA G-23 = MWRDGC 93) did not indicate a problem. Nevertheless, copper will be analyzed in more detail in the next Tier II evaluation.

Table 2.7 Parameters Not Meeting Illinois General Use Standards or Threatened

Parameter	Representative Lower Des Plaines River Sites Not Meeting General Use Standards	Comment on meeting the Secondary Contact and Indigenous Aquatic Life Standards
Copper	MWRDGC Sites (chronic & acute)[1)]	All sites meeting Illinois secondary use standard
Mercury	MWRDGC Sites (chronic & acute)[1)]	MWRDGC sites 92 - 95 also not meeting the secondary use standard
Fecal Coliform	All stations	No Illinois secondary use standard in force
pH	MWRDGC sites 94 & 95	Also not meeting Illinois secondary use standard
Dissolved Oxygen	All stations with exception of MWRDGC 95 (Interstate 55)	Only Stations G23 and MWRDGC 93 do not meet the secondary use standard
Zinc	All MWRDGC sites [1)] (IEPA measurements not available)	Only acute Illinois General use standard is met at all sites. **Illinois chronic standard is not met at all sites.** Federal chronic criterion is met at all sites.

[1)] MWRDGC sites measured total metals only.

Mercury. This metal has a very low standard (CMC= 2.6 μg/L, CCC = 1.3 μg/L, respectively) for total concentrations. Oddly, the Illinois secondary use indigenous aquatic life standard for total mercury is even less, 0.5 μg/L. The probability plots for mercury (Appendix B) show that most measurements at MWRDGC 92-95 and IEPA G-23 are below the detection limit of 0.1 μg/L. However, all MWRDGC sites have one to three measurements that exceed the standards. The reference site has only one measurement of 0.07 μg/L that is greatly below the standard. The compliance probability for the sites is given below

Reference site	> 99.8%	(Only one measurement)	
IEPA G-23	> 99.8%	(All measurements below detection limit)	
MWDDGC - 92	98% CMC	96% CCC	Upstream site CSSC
MWRDGC - 93	98% CMC	96% CCC	Brandon Dam pool
MWRDGC - 94	98% CMC	97 % CCC	Dresden Island pool
MWRDGC - 95	96% CMC	95% CCC	I-55 (Dresden Island pool)

Mercury is a problem that may have to be addressed by a TMDL study. However, before such study is initiated, analytical measurements with a lower detection limit should be conducted for several years. It is very difficult to estimate loading capacity and other variables of TMDL if a majority of measurements are reported as detection limit. Also, a significant part of the mercury load may be uncontrollable or difficult to control atmospheric emissions.

Fecal coliform bacteria. All sites indicated noncompliance with the Illinois General Use Standard for primary contact recreation. The probability level of compliance with the probabilistic standard of the maximum 10 % of samples in any 30 day period not exceeding 400/100 mL is given below. 10% allowable exceedance means 90% or more compliance.

The matter of fecal coliform compliance or noncompliance may be simplified by the new USEPA (2002) draft guidelines that specify Escherichia Colin as an indicator organism and link the magnitude of the standard to the risk of gastrointestinal disease to swimmers.

Fecal coliform compliance at monitored sites

	Compliance	
Reference site	85 %	Kankakee River
IEPA GI - 02	50%	Upstream site CSSC
MWRDGC 92	60 %	Upstream site CSSC
IEPA G - 23	50 %	Brandon Road Dam Pool (Joliet)
MWRDGC 93	50 %	Brandon Road Dam Pool (Joliet)
MWRDGC 94	20 %	Dresden Island Dam Pool
MWRDGC 95	50 %	Dresden Island Dam Pool

The attainability of the bacteriological standards and definition of uses and new risk based E. Coli standards for the Brandon and Dresden Island Pools is presented in Chapter 7.

Dissolved oxygen. Dissolved oxygen in the Brandon Pool of the Lower Des Plaines River frequently falls below the General Use Standard of 5 mg/L. The river is made of two impoundments that have a very low reaeration capacity. Removing the dams and improving in this way the reaeration is not possible because active navigation on the river is a protected beneficial use, based on the interpretation of the wording of the Clean Water Act. As a matter of fact, overflows over the Brandon Road Dam are the major source of DO in the Dresden island pool.

The standard of 5 mg /L DO is met with the following probabilities of compliance (note that on the probability distribution charts in Appendix B the compliance is assessed from right to left, i.e., a 20% reading on the probabilistic - proportion scale means 80% compliance):

	Compliance	
Reference site	99 %	Kankakee River
IEPA GI -02	60 %	Upstream site
MWRDGC 92	50 %	Upstream site
IEPA G 23	75 %	Brandon Road Dam Pool (Joliet)
MWRDGC 93	80 %	Brandon Road Dam Pool (Joliet)
MWRDGC 94	99 %	Dresden Island Dam Pool (Empress)
MWRDGC 95	>99.8 %	I-55 Dresden Island Dam Pool

A 99% compliance for the reference site may not be an acceptable compliance for the 5 mg/L "absolute" minimum standard specified by the Illinois General Use Standards. Analysis of the continuous monitoring of DO in Joliet on Brandon Pool (MWRDGC) and I-55 (Midwest Generation) will be done in the subsequent next Tier II analysis that will address the DO attainability in more detail.

Zinc. The compliance with the General Use chronic standard and federal criterion for zinc is presented below for the MWRDGC sites

Site	Illinois General Use Standard				Federal Criterion			
	Acute		Chronic		Acute		Chronic	
	µg/L	% Compliance	µg/L	% Compliance	µg/L	% Compliance	µg/L	% Compliance
MWRD GC 91	310.6	>99.8	55.6	75	297.5	>99.8	269.4	>99.8
MWRD GC 92	249.9	>99.8	44.8	45	239.4	>99.6	216.8	>99.8
MWRD GC 93	263.4	>99.8	47.2	50	252.3	>99.8	228.5	>99.8
MWRD GC 94	265.9	>99.8	47.2	40	254.7	>99.8	230.7	>99.8
MWRD GC 95	265.9	>99.8	47.7	52	254.7	>99.8	230.7	>99.8

It is clear that chronic General Use standard for zinc is not met and the excursions are significant. At some sites more than 50 percent of measured values do not comply with the standard. The question that must posed and answered is whether the chronic General Use standard is attainable. The data base did not contain measured values at the reference streams; therefore, Reason 1 of the UAA attainability cannot be reliably used. However, the reality of the standard should be reviewed by comparing it with the federal USEPA chronic criterion that is about 5 times greater and is attained at the measured sites. Therefore, it is not a question of attainability of the chronic General use standard that should be answered, it is the question of reality of the standard and its overprotectiveness.

Parameters Not Addressed by This Report

Several parameters and causes of impairment listed in the Illinois 303(d) list have not been addressed in this report.

Priority organics in the water column. Data on priority pollutants other than phenol and toxic metals were not provided. The State of Illinois has only a narrative standard that would require development of a numeric translator. Most of the federal criteria are for use of water for drinking and fish consumption. Priority organics in the sediment are addressed in Chapter 3.

Nutrients. Illinois does not have a numeric standard for nutrients. The federal draft criteria document provides only a ranking of the water bodies within the ecoregion and does not address the use impairment. Nitrate, a product of the nitrification process in the treatment plants and in the receiving water, has been increasing as shown on Figure 2.9. Figure 2.10 shows a corresponding decrease of the Total Kjeldahl Nitrogen (TKN) that is converted to nitrate in the nitrification process. Removal of nitrate from the effluents is possible by modifying the treatment plants to include nitrification/denitrification. Such processes are common in Europe and many US treatment plants (e.g., Brookfield, WI). Nitrate is approaching in the river the drinking water limit of 10 mg/L but not exceeding it. Because potable water use of the Lower Des Plaines River is not an existing use and no problems were encountered at the nearest site (Peoria) the problem was not analyzed further. Figure 2.12 presents the phosphorus concentration.

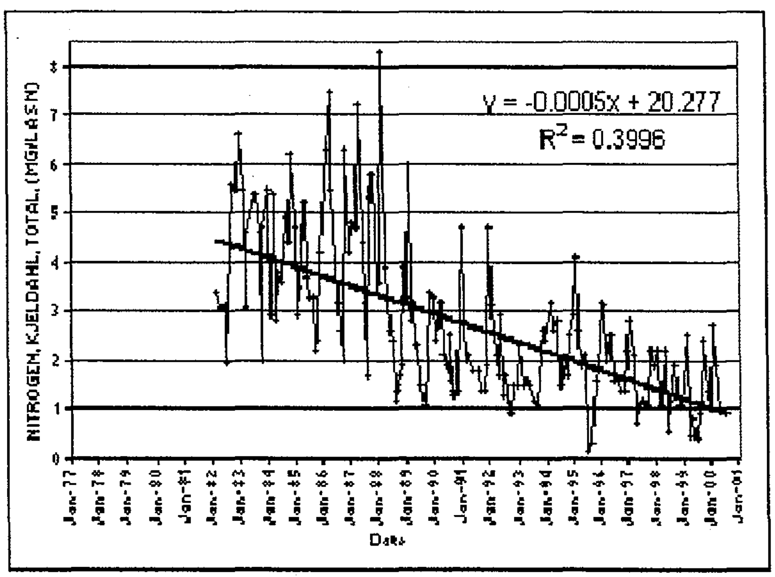

Figure 2.11 Historic Total Kjeldahl Nitrogen at G-23

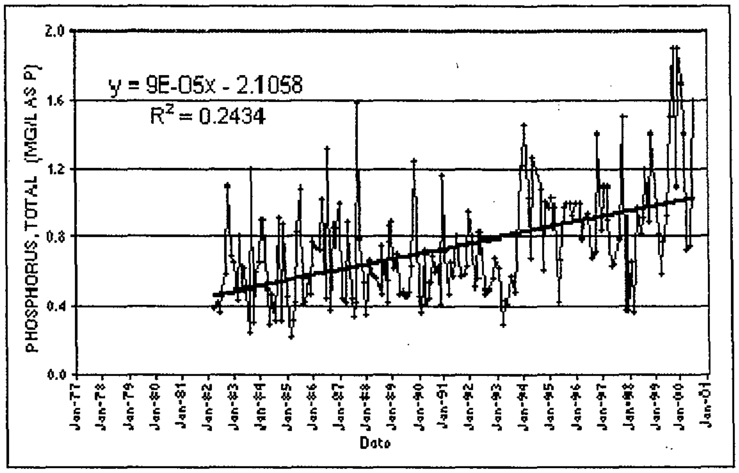

Figure 2.12 **Historic Phosphorus Concentrations at G-23.**

The nutrient level is exceeding the concentrations limiting eutrophication and eutrophication symptoms are evident (e.g., daily fluctuations of the DO concentrations, high turbidity partially due to algal infestation). The problem of daily fluctuation of dissolved oxygen, caused most likely by nutrient enrichment, will be discussed in the subsequent section of this chapter.

In spite of the absence of quantitative standards for nutrients, the problem can not be overlooked. However, most of the nutrient loads come from the upstream reaches of the Chicago Area Waterway System and should be addressed in the subsequent UAA.

Siltation and habitat alteration. These potential causes of use impairment will be addressed in subsequent Chapters 4 to 6 on biological impairment and the ecologic potential of the Des Plaines River.

Tier II Evaluation

Tier II e valuation follows the screening done in Tier I. Ammonium, copper, pathogens, dissolved oxygen and temperature were identified for further analysis. Analysis of pathogens (fecal colifoms and Escherichia Coli) is included in Chapter 7 where appropriate standards will be developed.

Ammonium[5]

Ammonium and Total Kjeldahl Nitrogen (a sum of total ammonium and organic nitrogen) have been declining in the last ten years (Figure 2.13). Apparently, a change in operation of the aeration equipment resulted in more nitrification. The obvious result was an increase of nitrate N which is a product of nitrification (Figure 2.10). Ammonium and organic nitrogen are measured together as Total Kjeldahl Nitrogen. Therefore, the trend of TKN on Figure 2.11 is similar to that of ammonium on Figure 2.13. Because ammonium is a part of the TKN analysis, the high ammonium concentration in winter of 2000 is mostly an outlier because it was not accompanied by a corresponding high TKN.

Ammonium is evaluated using both acute and chronic standards. The acute feferal criterion is a function of pH and applied to the instantaneous grab values. The chronic standard is a function of both temperature and pH, and is estimated using a 30 - day moving average of samples. Due to the fact that the number of samples taken by the agencies does not allow estimating 30 day moving average a direct estimation of ammonia concentrations compliance with the CCC standard is not possible.

In such situations, the USEPA allows use of the Monte Carlo methodology. The most important advantage of Monte Carlo modeling is compatibility with the water quality standards expressed in terms of allowable probability of exceedance. Monte Carlo modeling software is included in the USEPA models DYNTOX and its concept is described in Marr and Canale (1988). The methods and derived software allow time averaging by the moving average concept (4 or 30 days) and includes formulas, where needed, for calculations of site specific criteria for metals and ammonia. The US EPA's model QUAL 2E (downloadable from the US EPA watershed web site-www.epa.gov) has also Monte Carlo capabilities. Another Monte Carlo application with a more complex water quality transfer function was developed and published in a peer reviewed article by Novotny, Feizhou, and Wawrzyn (1994). Monte Carlo modeling was also suggested by the EPA researchers (Ambrose et al., 1988) as a recommended methodology for waste load allocation (and, hence, for TMDL).

The Monte Carlo analysis begins with the measured incomplete series of concentration values for the parameter of interest (e.g., ammonium). The term "incomplete" means that samples were not measured daily and there are large data gaps in the measured series. The Monte Carlo methodology substitutes missing data by computer simulation using the original probability distribution of the

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[5] The terms ammonium and ammonia are sometimes used interchangeably in the water quality standards literature. In this report ammonium refers either to the total ammonium (NH_4^+ and unionized NH_3) or to the ionized form. The term ammonia refers to the unionized toxic form, NH_3, which is a gas that can be dissolved in water.

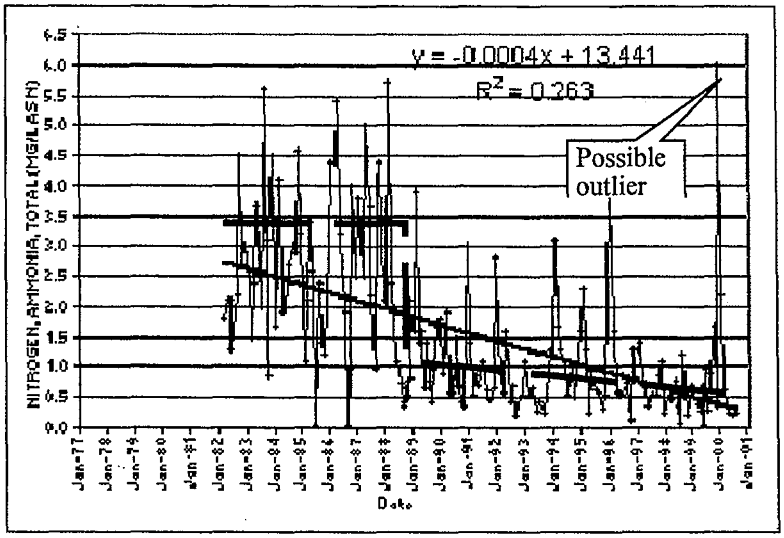


Figure 2.13 Historic Plot of Total Ammonium at G-23

incomplete series. In Monte Carlo modeling methodology a random number generated by a suitable computer program is transformed into a cumulative exceedance probability value, which is then applied to the probability distribution of the parameter(s) of interest, thus obtaining a value that is used as a substitute for the missing measured value. This process is repeated many times (on the order of several thousand). In this fashion, as a large number of data points become available, the series can be statistically evaluated, and the number of exceedences of the standard can be counted. The generated series of data has exactly the same probabilistic distribution as the measured incomplete data series. This series can then be averaged to obtain 30-day (or any other number of days such as four) mean values that can then be statistically analyzed for exceedences of the pertinent CCC-standard. A simple spread sheet model in the Excel environment was created by the AquaNova/Hey Associates researchers that calculated the data series of ammonium concentration, averaged them over a 30-days moving average windows, calculated the CCC standard for each 30 day period from average temperature and pH for the period and calculated a ratio of 30-day ammonium concentration divided by the CCC standard. The CCC standard was calculated by the equation taken from USEPA (1999) ambient water quality criteria for ammonium listed in the footnote of Table 2.1. In this case a ratio of less or equal to one signifies a compliance with the standard and greater than one is noncompliance, respectively.

The generated series of compliance ratios are plotted on Figures 2.14 (IEPA data) to 2.16 (MWRD data). Note that the simulated period is six years (1995-2001). This simulation for this period was recalculated several times to get an average number of exceedences in 3 years that was then used for evaluating the compliance.

The detailed analysis of the ammonium concentrations resulted in the following outcome and conclusions:

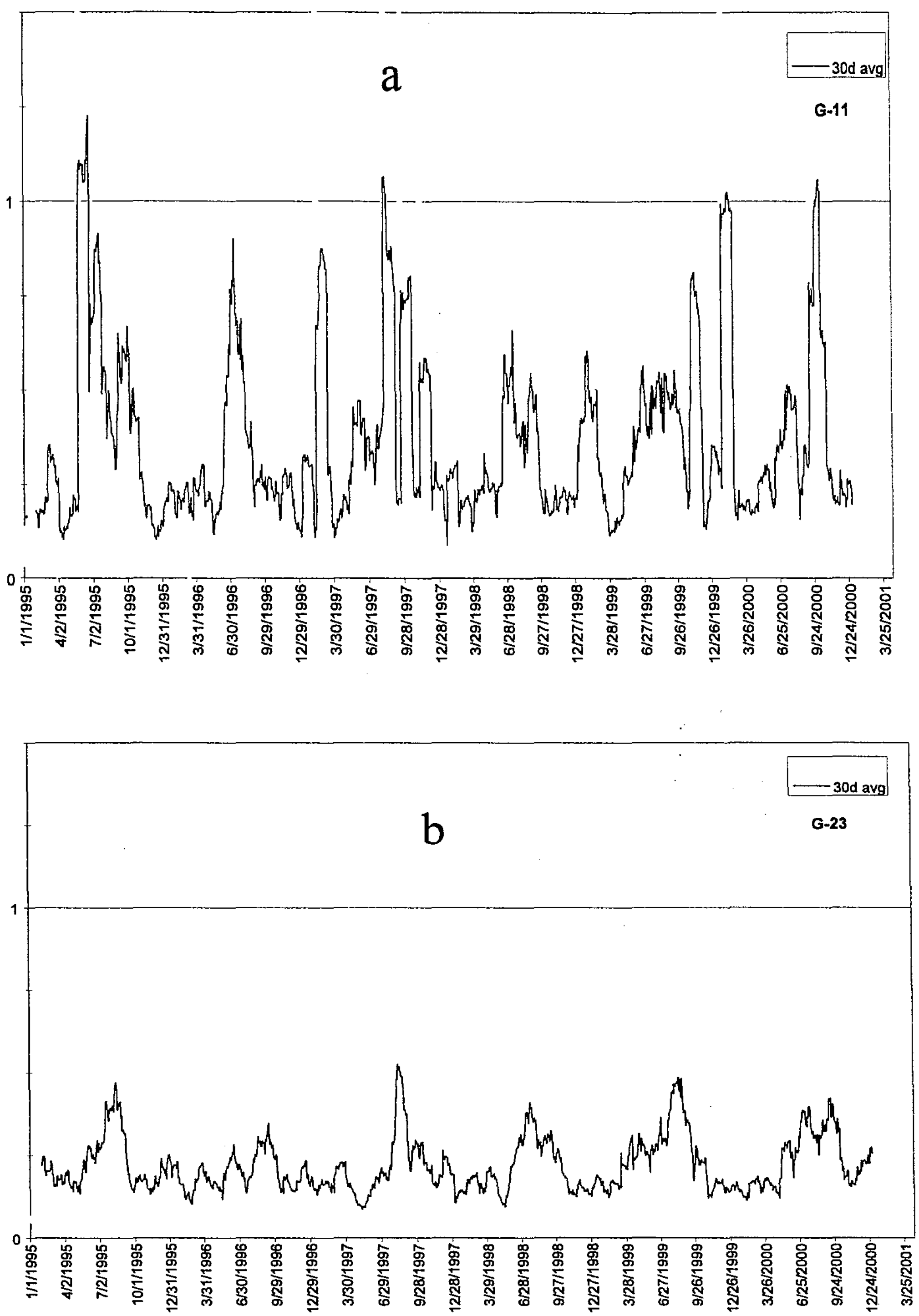


Figure 2.14 ab Compliance of ammonium concentrations with CCC 30-day moving average standard IEPA sites G-11(a) and G-23 (b)

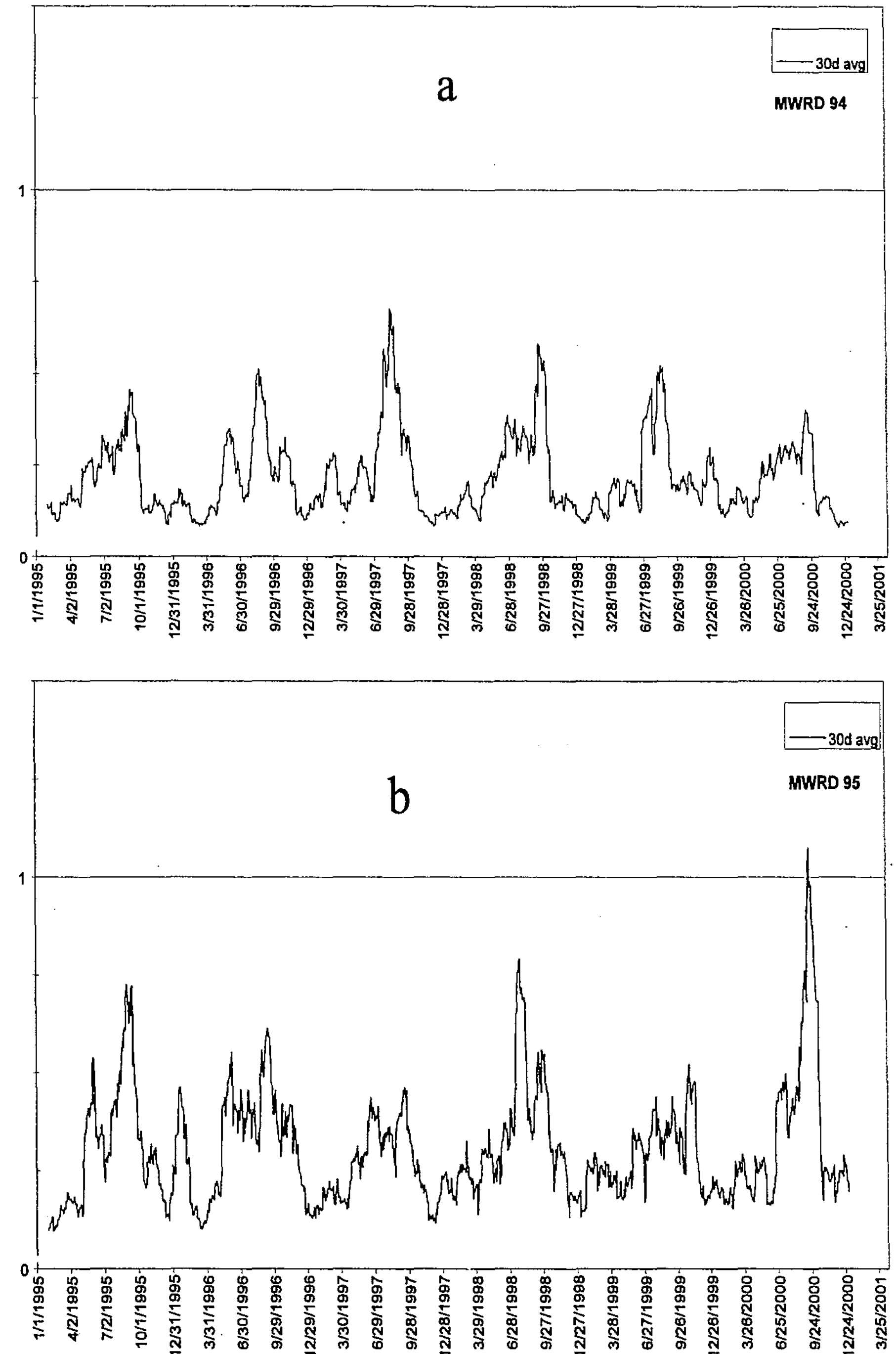
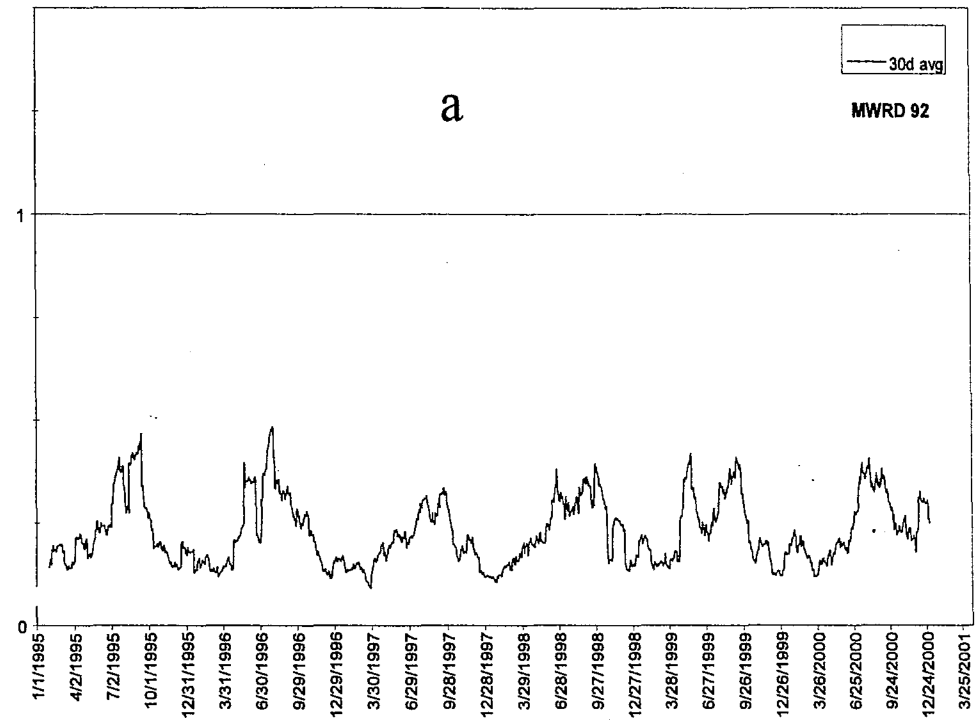


Figure 2.15ab Compliance of ammonium concentrations with CCC 30day-moving average standard. MWRDGC sites 92 (a) and 93 (b)



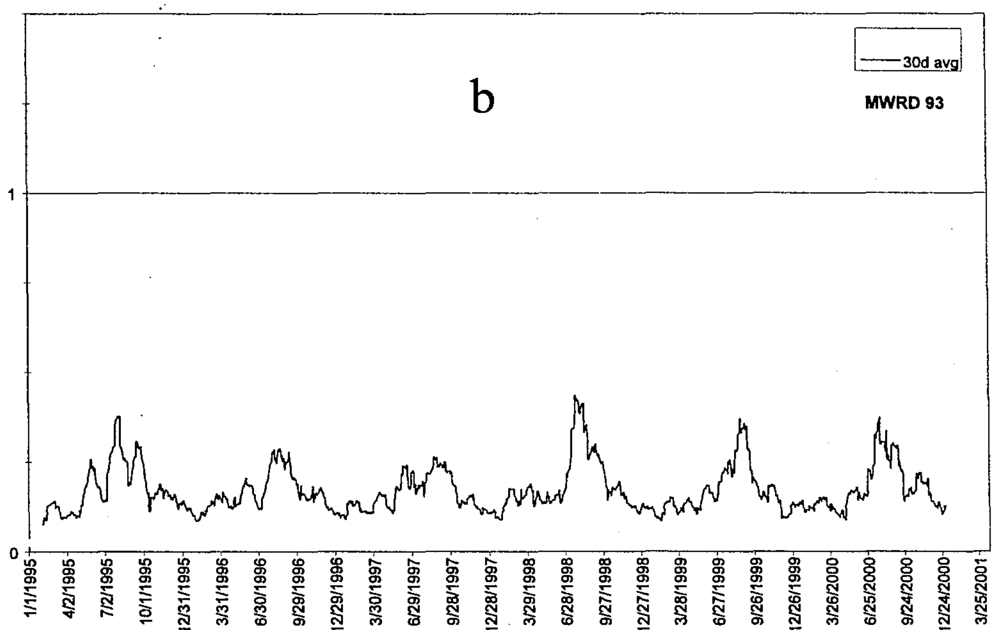


Figure 2.16 ab Compliance of ammonium concentrations with CCC 30day-moving average standard. MWRDGC sites 94 (a) and 95 (b)

Acute standard for Ammonium

In the Tier I, all measuring stations had a probability of compliance with the USEPA (1999) CMC criterion greater than 99.9%. It was found that the CMC standard is attained.

Chronic standard (30 day moving averages) - Monte Carlo simulation

Station	Number of exceedances in 3 years	
I EPA G-11 (upstream Des Plaines River)	0.5 (1 in six years)	small MOS
MWRDGC 91 (upstream Dees Plaines River)	0	Large MOS
MWRDGC 92 (upstream CSSC, Lockport)	0	Large MOS
I EPA G-23 (Brandon Road Pool, Joliet)	0	Large MOS
MWRDGC 93 (Brandon Pool Joliet)	0	Large MOS
MWRDGC 94 (Dresden I. Pool, Empress C.)	0	Large MOS
MWRDGC 95 (Dresden Isl. - I 55)	0.5 (1 in six years)	small MOS

The results of this analysis indicate that the chronic standard for ammonium would most likely be attained at all stations. The Margin of Safety would be large for all stations of the Lower Des Plaines River except MWRDGC 95 (I-55) where combination of higher pH caused by algal development and high temperature would result in a small MOS.

Copper

In the Tier I water body analysis, copper was identified as a parameter that did not meet the water quality standards at the locations on the Lower Des Plaines River analyzed by the MWRDGC while the IEPA analysis at the G-23 location indicated compliance. The difference of the analyses and sample collection might have been a partial problem. The monitoring at the IEPA station analyzed dissolved copper while the MWRDGC stations 93 (Brandon pool), 94 and 95 (Dresden pool) measured total copper concentrations. The IEPA analysis at the G-23 showed all measurements of dissolved copper below the detection and also below the dissolved copper CMC and CCC standards. MWRDGC 93 and 95 had borderline compliance. The acute CMC standard was fully met while the chronic CCC compliance was doubtful. Therefore, the analysis will be performed primarily at the MWRDGC station 94, where both standards were exceeded. Note that the once in 3 years allowable frequency of excursions for the CMC standards is equivalent to 99.8 % compliance. In the Tier I analysis of the chronic toxicity evaluation that requires four day averages, the assessment was only approximate and the CCC standard was compared with the 99.4 percentile concentration.
The more detailed Tier II analysis proceeded as follows:

1. The data were analyzed to reveal seasonal changes of the copper concentrations.
2. Sources of elevated copper were identified.
3. A water effect ratio was developed from IEPA data (both total and dissolved concentrations were analyzed) and applied to the MWRDGC data to obtain estimates of dissolved concentrations. These were then compared with the dissolved standard.
4. A modified standard was developed using USEPA's procedure for site specific standard development for the locally indigenous species.

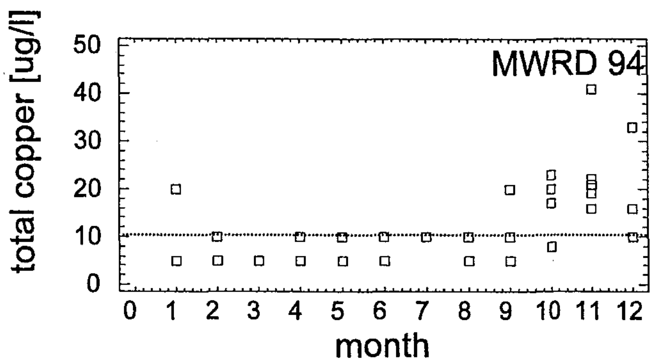


Figure 2.17 Monthly Variations of Copper at MWRDGC 94

5. Development of the water effect ratio, WER, based on the toxicity measurements in the water body and the laboratory water was suggested. The rationale behind this standard procedure is that river water may contain ligands that detoxify copper. Such ligands may not be present in the laboratory water in which the bioassays for copper toxicity were performed.
6. The final step was to estimate a simplified TMDL as a reduction of point and nonpoint copper discharges.

A full detailed report on copper analysis was submitted to the IEPA and stakeholders for evaluation and comments. The full report is included in Appendix C. The subsequent sections are a summary of the full report.

Seasonal Variations

Figure 2.17 shows the monthly variations of the total copper concentrations at the MWRDGC site 94. Most of the data were below the detection limits of 5 and 10 µg/L. Only in late fall and winter were higher concentrations measured. However, this pattern is specific only for the MWRDGC data collected over a two year period and has not been found in the long term sampling by IEPA at G-23.

Sources of Copper

Natural ecoregional sources. Copper is a common trace metal that is found in nature as a free metal (Cu^0), copper sulfide (CuS_2), chalcopyrite ($CuFeS_2$) and in other forms. It is measured in small concentrations in ground and surface waters. However, a study by Schonter and Novotny (1993) found that natural concentrations of copper in the reference water bodies located in the Milwaukee River watershed were typically less than 1µg/L. The analyses perfomed by the University of Wisconsin on this pristine watershed required ultra clean techniques.

Reference agricultural watersheds. In 1993 AquaNova study (Schonter and Novotny, 1993) concentrations of copper in nonurban reference watersheds were strongly correlated with the percent of the watershed in agriculture. The ranges of copper concentrations in reference watersheds that are not impacted by urbanization and had less than 60 % agricultural land use (more than 40 % forest and wetland) were found to be between 0.0025 to 0.116 µg/L. A reference watershed near Milwaukee, WI that was 70 % agricultural and about 3 % urban had copper concentrations

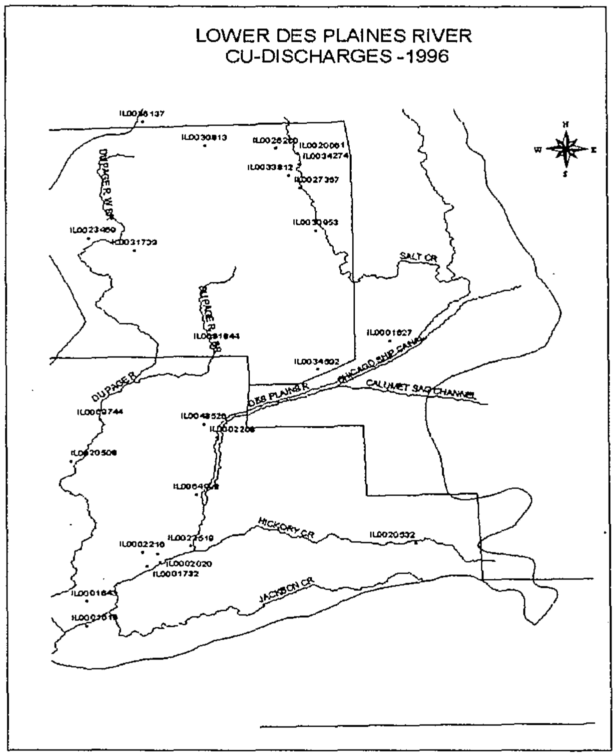


Figure 2.18 Permitted Point Sources of Copper in the Watershed of the Des Plaines River

comparable to those measured in the Lower Des Plaines River (10 to 40 µg/L). Concentrations of copper measured by the IEPA on the reference site of the Kankakee River at Momence were below the detection limit of 10 µg/L.

These relatively low concentrations of copper measured at nonurban reference watersheds effectively discount the possibility that the elevated copper concentrations measured in the Lower Des Plaines River would be of a natural origin (Reason 1 of the UAA regulations for modification of standards or use).

Urban sources. Urban sources of copper are numerous and include point and nonpoint sources. The National Urban Runoff Project study by the USEPA (1983) found urban stormwater runoff annual copper loading higher than effluent from secondary treatment plants. The sources can include metal corrosion (pipes, copper roofs), automobile emissions and wear out, use of copper based algicides, and a number of industrial sources such as paints, wood preservatives, and electroplating. Median copper concentration in a urban runoff in the NURP studies was 34 µg/L.
Figure 2.18 presents the registered point source of copper in the Des Plaines River and CSSC drainage areas. The map was obtained from the downloadable USEPA data base in BASINS, which also has a list of sources by name. However, the map shows primarily the sources that have copper mentioned in their permit. It is not implied in any way that these sources discharge excessive amounts of copper into the Des Plaines River.

The study by AquaNova International, Ltd. (Novotny et al., 1999) for the Water Environment Research Foundation found that winter use of deicing salts may contribute to elevated levels of toxic metals, including copper, during winter conditions. The salt itself contains copper. Novotny et al. (1999), Doner (1978) and Warren and Zimmerman (1994) documented that increasing concentration of chlorides (salinity) has a profound effect on the magnitude of the partitioning coefficient. Chloride concentrations found in urban runoff and streams after the application of deicing chemicals during winter can reduce the magnitude of the partitioning coefficient by several orders of magnitude. Consequently, metals can be leached from the soil adjacent to salted roads that have a higher metal content due to traffic and from metal laden sediments in urban detention ponds and streams. Donner (1978) found that increasing the Cl^- concentration in soil increased the rate of mobility of Ni^{++}, Cu^{++}, and Cd^{++} through soil. The increased mobility was related to the formation of chlorocomplexes and more dissolved metals in the soil environment. However, salting could be discounted as a source because the elevated copper concentration occurred in the October - December period during which (at least in October and November) salting is not practiced.

Relation to Flow

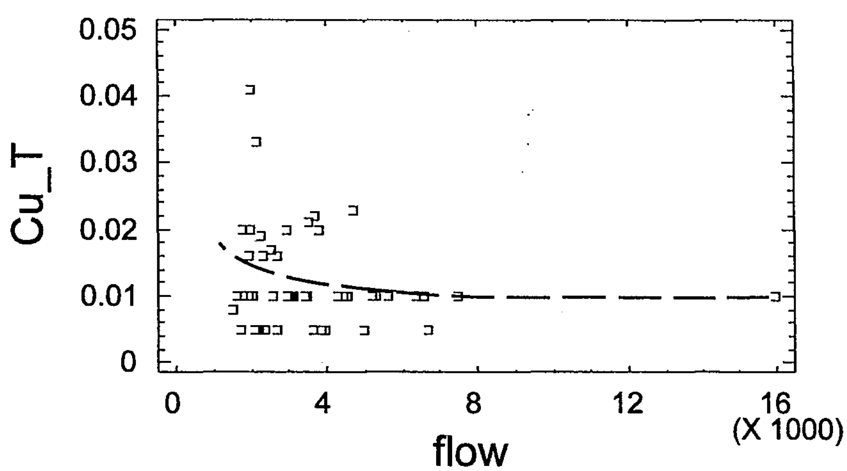


Figure 2.19 Plot of Copper Concentrations with Flow at MWRDGC 94 Monitoring Station

Figure 2.19 shows the copper concentrations at MWRDGC 94 plotted vs. flow. The largest concentrations occurred during low flow. This type of relationship is not typical for diffuse wet weather sources that would have the highest concentrations during wet weather larger flows. It resembles an effect of one or more point source discharges, which is most profound during dry weather conditions.

Water Effect Ratio: Estimation of Dissolved Copper

The strong affinity of fine sediments - primarily clay and organic particulates - to adsorb and make the pollutants biologically unavailable is considered by some as a partial water quality benefit of sediment discharges. The new USEPA water quality standards consider the effect of suspended sediment on the toxicity of metals (USEPA, 1994). Through sediment - dissolved fraction partitioning, the bioavailable fraction of toxic pollutants is reduced. For example, at concentrations of suspended sediment ranging from 15 to 50 mg/L, only about 25 to 30 % of copper would be available and toxic (Tischler and Hollander, 1994).

The IEPA has analyzed both dissolved and total concentrations of copper while the MWRDGC measured only total concentrations. As stated before, the total and dissolved copper measurements attained the standard and met the Illinois General Use. In the first step of this detailed assessment, dissolved concentrations are estimated from total concentration for the MWRDGC data. By developing a WER based on the correlation with the suspended solids and COD, both contain possible ligands that may precipitate copper.

The ratio between the dissolved concentration, c_D, and total concentration, c_T, is described by the partitioning theory:

$$\frac{c_D}{c_T} = \frac{1}{1+\Pi_{SS}\, c_{SS}}$$

where $_{SS}$ is the partition coefficient [L/mg] and c_{SS} is the concentration of suspended solids [mg/l]. Figure 2.20 shows the relationship between the dissolved-to-total ratio and concentration of suspended solids (SS). Data showing inconsistencies (c_D>c_T) or detection limits were eliminated from the analyses.

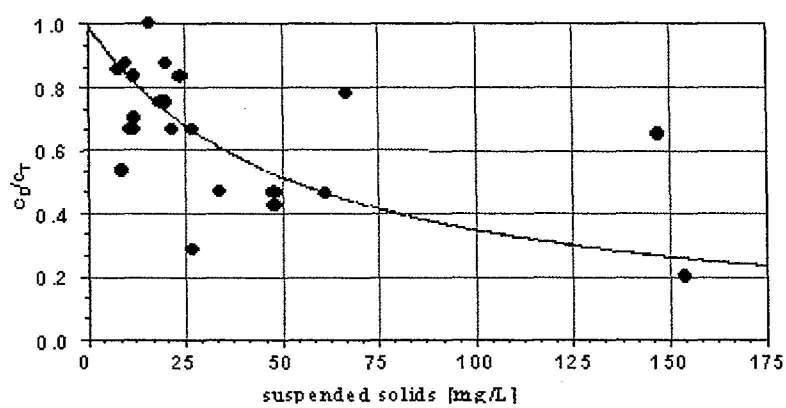


Figure 2.20 Changes in c_D/c_T with Changes in Suspended Solids Concentration: Partitioning Theory Fit

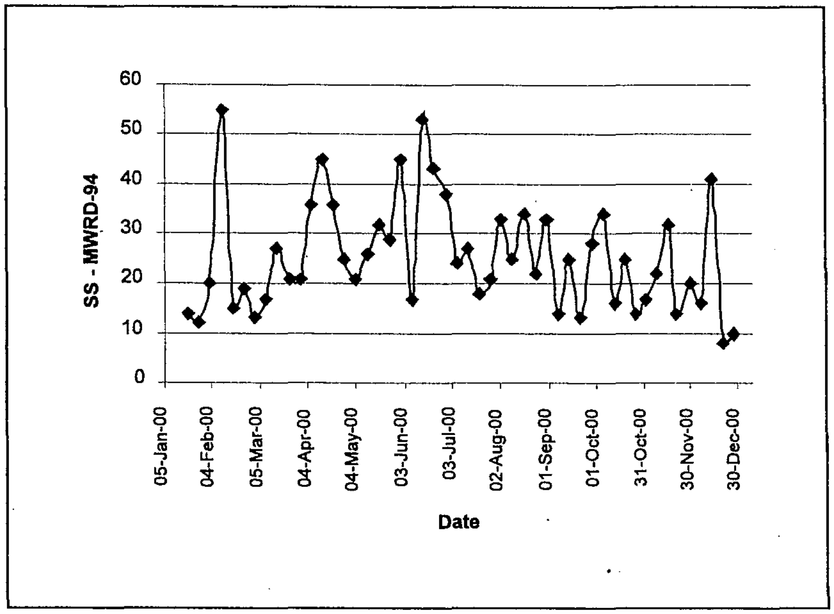


Figure 2.21 Suspended Solids Concentration at the MWRDGC Station 94 (Empress Casino)

The correlation coefficient was sufficient. However, there are only a few measurements for high concentrations of suspended solids and the spread is quite significant. From the analysis the partitioning coefficient, $\Pi = 0.01896$ L/mg x 10^6 Kg/mg $\approx$ 19,000 L/Kg.

In addition to suspended solids the correlation was also conducted for SS and COD to account for the fact that organic particulates also immobilize metals. However, COD was found to be strongly correlated to suspended solids, therefore; COD was dropped from the relationship.

The variation of suspended solids in the Des Plaines River is significant because the sediments are continuously being resuspended by barge traffic. Figure 2.21 is a plot of suspended solids in the Des Plaines River. A high concentration spike is a result of a barge tow transient resuspension of the bottom sediments (see the discussion below on the effects of barge traffic). This range is most common.

Sediment as a Source of Copper

Table 2.8 contains the sediment copper concentration data for the Des Plaines River at Brandon Pool (Rm. 290.5), Dresden Island Pool at Rm. 285 (1 mile downstream of Brandon Road Dam), Dresden Island Pool at Rm 278 (I-55 Bridge), and the Reference Kankakee River at I-55 near Wilmington.

The data were provided by the MWRDGC. Only the data between 1994 and 2000 were considered[6]. All measurements were made in the month of October. Upon investigating the average copper concentrations of the sediment it becomes evident that is a significant difference of copper (metal) contamination in the Dresden Pool between the Rm 285 and Rm 278. The sediment concentration of copper between these two locations doubles. This is in agreement with the water column evaluation. One explanation is that the sediment at RM 285 has a coarser texture and less volatile solids than at RM 290.5 and 278, indicating that the sediment has a less adsorbing capacity for copper.

IEPA classified the sediments in the state waters based on a classification contained in Short (1997). The categories were nonelevated, elevated and highly elevated. Based on this comparative classification the copper content of the sediments in the Lower des Plaines River would be classified as either uncontaminated by copper (Dresden Pool at RM 285) or mildly contaminated (Brandon Pool and Dresden Pool at RM 278).

The key parameter that defines sediment contamination, besides the total concentration of the pollutant, is the pollutant concentration in the pore water of the sediment. The pore water reflects the toxicity of the sediment because the fraction of the particulate pollutant is considered as not being toxic (DiToro and DeRosa, 1995; DiToro, 2000). This is analogous to the concept of WER introduced in the preceding section that specifies that the dissolved concentration of the metal in water is toxic while the particulate metal is not. In addition, a judgement can be made as to whether or not the contaminated sediment is a source or sediment is a sink of the copper.

Table 2.8 Sediment Characteristics and Contamination by Copper (1996 ñ 2000)

Location	Total solids %	Total Volatile Solids %	Total Copper mg/Kg	Pollution Classificatio n
Brandon Pool (RM 290.5)				
Average	65.9	7.8	61.0	elevated
Range	67.1 - 71.1	4.9 – 12.1	57 - 66	
Upper Dresden (RM 285)				
Average	68.2	5.58	33.6	non-elevated
Range	55.8 – 77.8	3.4 – 8.0	23 - 51	
Lower Dresden (RM 278)				
Average	42.16	7.3	94.6	elevated
Range	40.5 – 66.1	44. – 11.3	44 - 158	
Kankakee R., (Wilmington)				
Average	NA	NA	21.7	non-elevated
Range			18 – 25	

[6] Chapter 3 has a detailed evaluation of sediment contamination using all available data.

Copper can be released from the sediment by

- Convection of pore water into the water column by groundwater discharge
- Diffusion if the pore water is much greater than the water column concentration
- Scouring of the contaminated sediment (e.g., by barge traffic)

Pore water concentrations were not measured but could be calculated by the same partitioning concept. For sediment

$$C_{pw} = \frac{C_T}{\theta + \Pi m_{ss}}$$

where C_{pw} is the dissolved copper concentration in the pore water, C_{Ts} is the total copper concentration in the sediment, θ is the porosity or water content of the sediment. m_{ss} is the solids content of the sediment in Kg/L and Π is the partitioning coefficient in L/Kg. Porosity was estimated from the percent weight of the solids in the sediment and average density.

Ambrose (1999) presented a statistical equation that relates water and sediment partitioning coefficients as

$$\text{Mean log } \Pi \text{ sediment} = 1.418 \text{ (mean log } \Pi \text{ suspended sediment)} - 3.18$$

The calculated pore water concentrations of copper then were

	Pore water concentration
Brandon Pool	0.079 mg/L
Upper Dresden Island Pool	0.044 mg/L
Lower Dresden Island Pool	0.122 mg/L

These pore water concentrations are significantly greater than the water column concentrations. Many water column concentrations were below the detection limits of 0.01 mg/L and 0.005 mg/L, respectively. By mass balance calculation it was found that 99.9 % of copper in the sediment is particulate and immobilized and only about 0.1 % is contained in pore water.

It is now possible to ascertain the approximate magnitude of the copper fluxes. The three possible mechanisms of copper release from sediments were listed above. The first possible route can be discounted because the water level in river impoundment is almost always above the surrounding groundwater table; therefore, the water flux through the sediment layer is downwards (the impoundments are recharging groundwater). Diffusion of dissolved copper from sediment pore water is likely but it may be counterbalanced by the downward convective flux of the river into groundwater. Furthermore, almost all copper is contained in the particulate fraction. This may leave the scour of the bottom sediments by barge traffic as the only major mechanism of enrichment of the Lower Des Plaines River by pollutants from the sediment.

Bhownik, et al. (1981) studied the effect of barge traffic on resuspension of sediment and concluded that:

- Tow passage increases suspended sediment concentrations.
- The increase in concentration is greater in channel border areas than in the navigational channel.
- The increase is more significant when the ambient suspended sediment concentration is low.
- The concentration is transient and may last 60 to 90 minutes.

In the absence of extensive modeling and monitoring data it was not possible to accurately assess the impact of barge traffic on resuspension of copper (and other pollutants) from sediments in the Brandon and Dresden Island pools. Studies by Bhownik, Soong and Bogner (1989) in the Ohio River and Bhowmik, Lee, Bogner and Fitzpatrick (1981) in the Upper Illinois River showed there was a significant but very transient resuspension of sediments during barge tow passage. The increases lasted between a few minutes and ten minutes, at most. Typically, sediment concentrations increased during the barge tow passage by as much as 90 mg/L but the concentration subsided to its pre-passage value in 10 minutes after the passage. Also the work by Butts and Shackleford (1992) on the Upper Illinois River did not find significant differences in sediment concentrations with and without traffic.

Due to the difference in the partitioning coefficients in water and in the sediment, more copper can be adsorbed on the sediment particles in water than in sediment. Therefore, although the total water column copper concentration may be slightly increased during the barge tow passage, the released sediment may scavenge the copper from the dissolved pool in the water and take it back into the sediment layer during resettling. Upon resettling, a part of the resettled pollutant will be released into the pore water. During resupension of sediment by barge tow traffic, possible scavenging of metals and hydrophilic priority organics by the resuspended sediment and subsequent resettlement has either no or a slightly beneficial effect on toxic concentrations of these pollutants in the water column.

Comparison with Site Specific Standard

The acute and chronic toxicity standards have been calculated according to IEPA guidelines included in Table 2.1. Two approaches were used in this study to ascertain compliance with the current Illinois General Use and Secondary Contact and Indigenous Aquatic Life uses. In Tier I, standards were calculated using average hardness for the site and total metal concentrations. These calculated standards are shown in Table 2..4. The Illinois Environmental Protection Agency in the draft document of implementation of water quality standards requires that the standard be calculated using the sample hardness. In the Tier II analysis, the total concentrations were converted by WER to estimate of dissolved concentrations and compared with the IEPA standards.

Alternative 1 ñ Standards Calculated for Average Hardness

Tables 2.9 and 2.10 show probabilities of compliance with the acute and chronic toxicity criteria using Alternative 1 – Average Hardness. The chronic toxicity standard is defined for 99.4%. The acute toxicity does not seem to be an issue (Table 2.10). Table 2.9 shows the chronic toxicity standard would be exceeded in all sites, regardless of the regression function used. The total concentrations were converted to their dissolved fractions by the water effect ratios related to the suspended solids documented in the preceding section on Water Effect Ratio: Estimation of Dissolved Copper.

Table 2.9 Probability of Compliance with the Chronic Toxicity Standard for Copper in MWRD sites [%], Assuming Log-normal Distribution

Method	91	92	93	94	95
Linear regression	99.363	99.168	99.274	98.761	98.944
Partitioning theory	99.243	98.437	99.173	98.688	98.629

Table 2.10 Probability of Compliance with the Acute Toxicity Standard for Copper in MWRDGC Sites [%], Assuming Log-normal Distribution

Method	91	92	93	94	95
Linear regression	99.960	99.951	99.953	99.902	99.919
Partitioning theory	99.937	99.866	99.943	99.891	99.878

Sites 91 and 92 are upstream sites (Des Plains River - 91 and Lockport CSSC - 92) are used only as an information of upstream situation.

All evaluated sites (92, 93, and 94) met the acute toxicity standard when WER partitioning was considered. Sites 92, 94 and 95 may still exceed the chronic toxicity standard. Although site 92 is not part of the Lower Des Plaines system, its proximity and dominant impact is indisputable. This site exhibits the most dramatic improvement due to application of WER and conversion of total concentrations to dissolved concentrations.

An increase of copper concentrations occurs between sites 93 and 94 exhibited by the decrease of the probability of compliance between the sites. Site 93 is in downtown Joliet in the Brandon pool, Site 94 is at the Empress Casino in the Dresden Island pool. There is also a decrease of the concentrations (exhibited by a small increase of the probability of compliance) between Site 92 at Lockport and 93 in Joliet. This may be attributed to a mild diluting effect of the Des Plaines River when it joins the flow from the CSSC. A recovery of the probability of compliance between the sites MWRDGC sites 94 (Empress Casino) and 95 (I-55) was also noted.

Alternative 2 ñ Standards Calculated for Each Sample

The draft Illinois EPA water quality standard's guidelines require that the standard is calculated for the harness of the sample and not for the overall average hardness of the site. A research by Bartošová and Novotny (2000) documented that the differences between the compliance of a standard based on the average hardness and standard determined for each sample from the sample hardness are not great. Determining compliance statistics for sample based standards requires a modified statistical analysis outlined herein as follows:

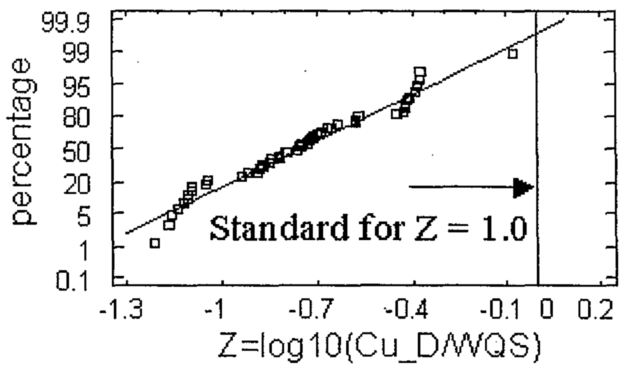


Figure 2.22 Probability plot for copper concentrations normalized by sample standard

For each sample, denoted as i in the sequence of samples:

- Calculate the standard using the hardness of the sample - WQS(i)
- Calculate the WER based on the suspended solids of the sample - WER(i)
- Calculate the dissolved concentration - CD(i) = CT(i) x WER where CT is the total concentration
- Calculate a new statistical variable - Z(i) = CD(i)/WQS(i)

For the sample being in compliance with the standard, Z is less or equal to 1.0. The variables Z were then statistically analyzed using normal and log-normal probability distributions. In this concept, the normalized standard for Z is 1.0 because all concentrations were divided by the sample standard calculated from the sample hardness. The respective limits of 99.8% for acute (CMC) evaluation and 99.4% for chronic evaluation are then applied in the way as for actual concentrations in Alternative 1. This concept is shown on Figure 2.22. The probabilities of compliance for all analyzed sections are in Appendix. C.

Tables 2.11 and 2.12 present the probabilities of compliance determined by the Alternative 2.

Table 2.11 Probability of Compliance with the Chronic Toxicity Standard for Copper in MWRDGC sites [%], Assuming Log-normal Distribution.

Method	91 upstream control site	92 upstream control site	93	94	95
Linear regression	99.184	98.321	98.878	98.263	98.556
Partitioning theory	99.243	97.591	98.856	98.075	98.172

Table 2.12 Probability of Compliance with the acute toxicity standard for copper in MWRDGC sites [%], assuming log-normal distribution

Method	91	92	93	94	95
Linear regression	99.934	99.803	99.892	99.807	99.850
Partitioning theory	99.932	99.690	99.893	99.771	99.780

The compliance probability, using the standard for each sample has not improved, although the differences are less than 1 %, commensurable with the results of the work by Bartošová and Novotny (2000). This proves that Alternative 1 methodology is adequate for screening and preliminary water body assessment. Since Alternative 2 is a methodology preferred and required by the Illinois EPA, the results in Table 2.10 and 2.11 will be considered and this methodology will be used for further assessment.

The results confirm compliance with the acute toxicity standards (Table 2.11) because all compliance probabilities were at or better than 99.8 %. The probability of compliance with the chronic standard (Table 2.10) has imroved; however, due to the incomplete data series (samples are taken in weekly intervals), it is not possible to arrive at an exact evaluation because such evaluation would require four days averaging of daily samples. Similarly to Alternative 1 evaluation it could be concluded; however, that the MWRDGC site 93, 94, and 95 data may not meet the compliance criterion for the chronic toxicity based on the USEPA frequency and duration (probability) for priority pollutants.

Site Specific Standards

The panel of experts on the biological subcommittee and the AquaNova/Hey Associates aquatic ecology experts developed a list of organisms that would be indigenous to northern Illinois rivers. The list was developed from the latest draft criteria document for copper (Great Lakes Environmental Center, 2001). The final set, plotted on Figure 2.23, contained 40 Genus Mean Acute Values (GMAV).

The procedure described in Appendix C followed the USEPA guidelines for developing site specific criteria in order to calculate the acute criterion of maximum concentration (CMC) and is also shown on Figure 2.23.

The final acute value (FAV) was determined as concentrations yielding 5% protection GMAV. The acute toxicity criterion is calculated as CMC = α*FAV, where the α multiplier corrects the FAV values derived from 50% lethality value LC50 to those that would correspond to a threshold-lethal (near zero mortality) effective concentration (USEPA, 1991). The recommended value for this procedure is $\alpha = 0.5$. The site specific criteria for individual sites are given in Table 2.13. The last column contains the criteria calculated from the standing formula (Table 2.1) of USEPA criteria and IEPA standard for metals. Daphnia magna is the most sensitive indigenous species that drives the magnitude of the standard. The standards estimated from the new USEPA guidelines are somewhat more stringent than the standing General Use standard. Table 2.14 then presents probabilistic compliance with the site specific standard. It should be noted that the methodology and criteria presented in the report by the Great Lakes Environmental Center (2001) is only draft guidance.

Table 2.13 Site-specific Standards for Acute Copper Toxicity

Site	Average hardness [mg $CaCO_3$/l]	FAV Gumbel	CMC Gumbel	CMC EPA
Reference (Kankakee)	294	81.43	40.72	46.93
IEPA - G-11	285	79.08	39.54	45.57
IEPA - G-02	231	64.97	32.49	37.44
IEPA - G-23	239	66.97	33.49	38.60
MWRDGC91	301	83.34	41.67	48.03
MWRDGC92	233	65.46	32.73	37.73
MWRDGC93	248	69.37	34.69	39.98
MWRDGC94	250	70.11	35.06	40.41
MWRDGC95	246	69.06	34.53	39.80
USGS Riverside	267	74.54	37.27	42.96
USGS Romeoville	210	59.51	29.76	34.30

Table 2.14 Probability of Compliance with the Acute Toxicity Standard for Copper in MWRDGC sites [%], Assuming Log-normal Distribution. Site Specific Standards

Method	91	92	93	94	95
Partitioning theory	99.86	99.70	99.87	99.76	99.74

All sampling sites meet the more stringent CMC (acute) site specific toxicity standard. However, the site specific standard, based on the indigenous aquatic biota would not change the conclusions on attainability of the chronic criterion.

Recalculation of the total concentrations of copper to their dissolved equivalents did not completely resolve the problem that the CCC standard is not met at the 99.4 % confidence level. This is shown on Figures 2.24 and 2.25.

TMDL Issues for Copper

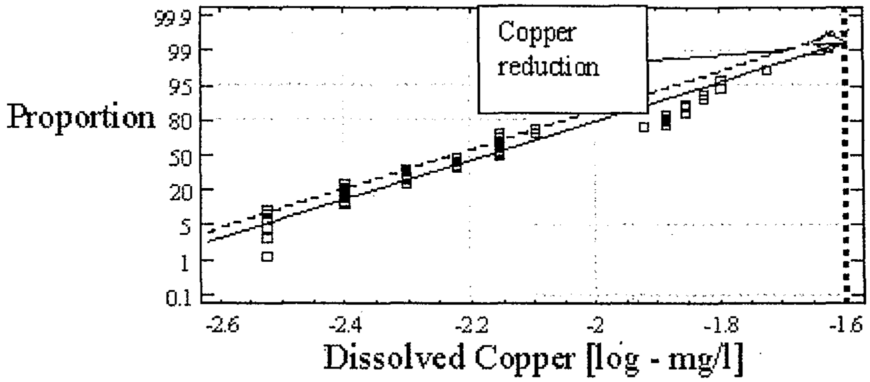


Figure 2.24 Probability Plot of Dissolved Copper at MWRDGC 94 Calculated by WER Related to Total Suspended Solids. Standard Estimated from Average Hardness of All Samples

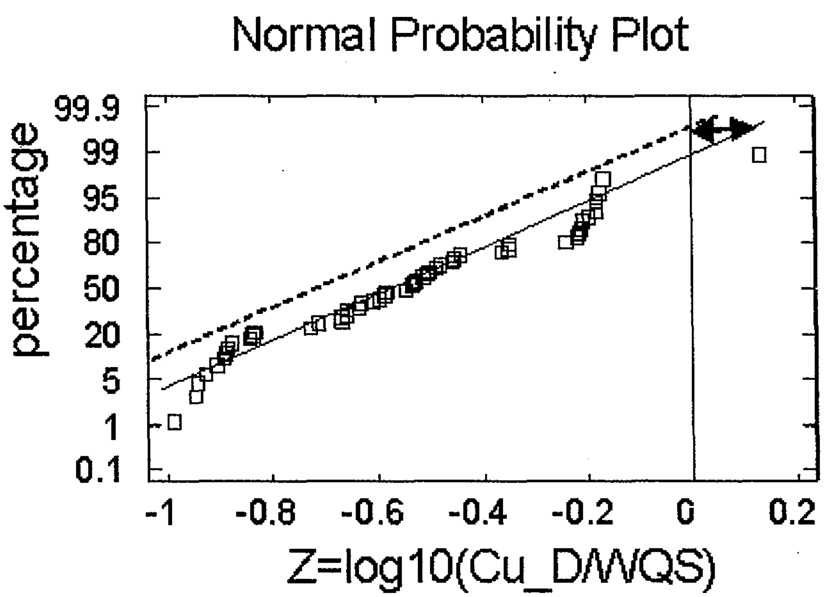


Figure 2.25 Analysis of the Calculated Dissolved Copper Using a Standard Estimated from Hardness of the Sample

The TMDL can then be estimated approximately by drawing a line parallel to the line of the best fit that would intercept the decision point (a point of intercept of the 99.4 % probability coordinate line with the vertical line denoting the CCC standard). The needed reduction on the logarithmic scale was -0.05, which corresponds to the required percent reduction of copper concentrations (TMDL) of

11%. For the sample based standard in the Alternative 2 and dissolved concentration, the required logarithmic reduction is – 0.08, or 12%.

The TMDL is expressed as a percent reduction of copper loads. Before implementing such restriction on the dischargers IEPA should consider developing a WER based on the difference between the river (effluent) toxicity of copper and toxicity of similar concentrations in the laboratory water from which USEPA developed the copper standard. The procedure for the development of the toxicity based WER is described in USEPA (2001). The effect of such WER was documented in the TMDL for the NY-NJ Harbor prepared by the USEPA Region 2 (1994). In the study, WER was analyzed and determined for the estuary. The estimated value of WER for copper was 1.5, which allowed increasing the copper standard by 50%.

Summary and Conclusions - Copper

The detailed report on compliance of MWRDGC data on total copper (Appendix C) then concluded:

- The historic plots of copper concentrations at the MWRDGC sites indicate that higher concentrations occur during the late fall and early winter months. This pattern is not repeated at the IEPA sites and may be coincidental. Low temperature and possible salinity increases have an adverse effect on copper binding and immobilization in the sediments, thus a possible speculative cause of increased concentrations could be a release of copper from the sediments and leaching by salt laden runoff from soils and urban/industrial sites. Other point and nonpoint sources may also be responsible but their impact is not known.

- The concentrations measured in the reference streams were below the detection limit and below the standard; therefore, natural and background causes of the elevated copper concentrations cannot be suspected.

- The detailed analysis confirmed compliance of copper concentrations in the Lower Des Plaines River with the CMC (acute) toxicity standard at the compliance level at or better than 99.8 %.

- Analysis of the water column and sediment copper concentrations indicate a possible source of copper between the MWRDGC water quality monitoring stations 93 (Joliet, Brandon Pool) and 94 (Dresden Island Pool, Empress Casino) and between Upper Dresden Island (RM 285) and Lower Dresden Island (RM 278) Pool sediment sampling points.

- The effect of barge tow traffic on copper concentrations is not great and is transient.

- Sediment appears to be mildly contaminated by copper. A more detailed study of sediment contamination would be needed before a recommendation for dredging of the sediment can be made. The next Chapter 3 addresses the issue of sediment contamination in a more comprehensive manner.

- Inclusion of the WER and conversion of the total concentrations into dissolved concentrations at the MWRDGC sites substantially increased the level of compliance with the CCC standard.

- A 10 to 12% reduction of the copper concentrations would be needed to meet the CCC standard at the MWRDGC site 94 (the critical site) at the 99.4 % compliance level. However, due to uncertainties with the methodology for CCC compliance that requires daily sampling, the percent reduction has a high degree of uncertainty and would be challengeable. The compliance percentage lies within the margin of error and uncertainty associated with the methodology. Also, the sources are not known from this analysis.

- A toxicity based WER performed according to the USEPA (2001) guidelines could result in an increase of the site specific copper standard for the Lower Des Plaines River and in compliance.

- The agencies should consider using clean analytical methodologies that would decrease the copper detection limit to or less than 1 g/L.

Based on the above summary points, the AquaNova/Hey Associates team concludes that

- The Lower Des Plaines River complies with the CMC (acute) standard based on the current interpretations of the water quality regulations that allow use of the WER and dissolved concentrations of the metal.

- A modification of the CCC evaluation methodology on the part of the USEPA is needed before an accurate final judgement is made whether or not the Lower Des Plaines River is in compliance with the CCC standard. At present, a possible probability excursion at the 99.4 percentile compliance criterion detected at the MWRDGC monitoring sites 93, 94 and 95 are marginal and certainly within a gray zone of knowledge. The required compliance of 99.4. % for the CCC standard is only a guidance that is not enforced by the USEPA nor by the IEPA. The problem of uncertainty with the frequency component of the CCC standard could be resolved by development of the toxicity based WER for the river that would lead to an increased site specific standard; therefore, development of toxicity based WER is recommended.

- Reasonable probability exists that with application of the toxicity based WER, compliance with the CCC standard may be achieved. Even current compliance with the CCC standard at the 98 % compliance level results in a very small risk, r 0.1 %, resulting in only daphnia being adversely but not lethally affected. Consequently, development of the TMDL allocation and assessment of a wide spread socio-economic impact (Reason 6) resulting from a possible watershed wide across the board 10 % reduction of copper loads is not recommended at this time.

- However, IEPA should be paying increased attention to the copper problem by promoting best management practices that would reduce copper inputs from urban and industrial runoff and from point sources discharging copper.

Zinc

Based on the comparison with the federal criterion we believe that the Illinois chronic standard is unnecessarily overprotective and unattainable. To meet the standard, zinc concentrations would have to be reduced by 70 to 90% which would require a very rigorous TMDL study. Because most of the metal comes from the urban nonpoint sources, such removals with the present best management practices may not be attainable. Before the zinc pollution is listed on the 303(d) list and a TMDL study is ordered, the IEPA and the Illinois Pollution Control Board should address the reality of the standard and reconcile the difference between the federal chronic criterion and the Illinois chronic standard.

The federal EPA chronic zinc criterion was met at the compliance level greater than 99.8%.

Dissolved Oxygen

Problems with Low DO

Dissolved oxygen adversely impacts the integrity of a receiving water body in several ways:

1. Low dissolved oxygen concentrations in water are toxic to fish, exhibiting both lethal and chronic effects.

    a. Low, longer duration DO concentrations inhibit growth and reproduction (chronic toxicity)
    b. Very low DO levels cause fish kills (acute toxicity).

    The toxic levels to fish species are different for cold and warm water fish. Cold water fish species require more protection (higher DO concentrations) than warm water species.

2. Low dissolved oxygen in the water column may change the upper sediment layer from aerobic to anaerobic (typically, lower sediment layers are devoid of oxygen). This changes solubility of some compounds and allows a release into the water column. Examples include ammonium/ammonia, phosphates, metals, and hydrogen sulphide. Anoxic or anaerobic upper sediment layers will cause a loss of aerobic benthic vertebrates that are important components of the food chain. Low DO concentration in the bottom substrate are also detrimental to spawning.

3. A complete loss of DO in water and/or all sediments changes the water body and sediment color to black, which is caused by sulphate reducing bacteria, resulting in the emission of methane and odorous hydrogen sulphide.

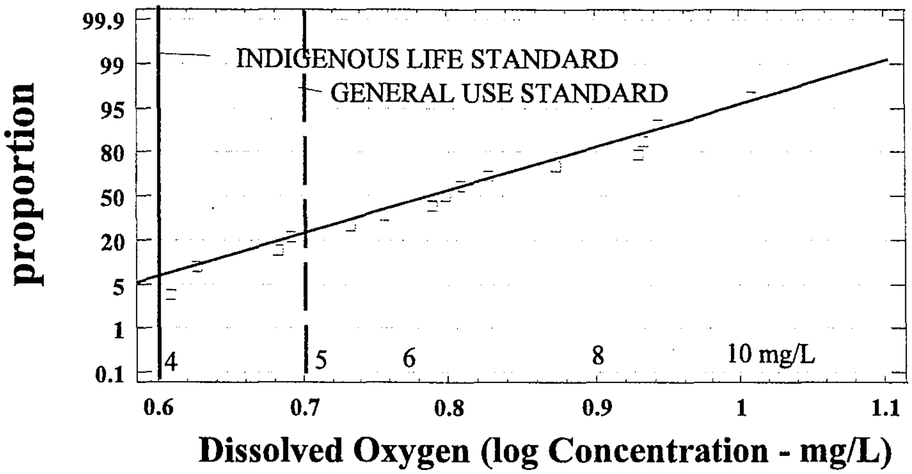


Figure 2.26 Example of Statistical Plotting of DO Concentration at the IEPA G-23 Sampling Point (Joliet)

Dissolved oxygen is the key parameter for determining the attainment of the designated use. The DO levels are affected by the discharges of the biodegradable organic matter from point and nonpoint sources, atmospheric reaeration, sediment oxygen demand, nitrification of ammonium and organic nitrogen, temperature and by algal photosynthesis and respiration (Thomann and Mueller, 1987).

The current DO criteria were presented in Table 2.1 and the history of the standard will be further elaborated in Chapter 7 where a possible modification of the standard will be proposed. The standards were derived from the Illinois Water Quality Regulations (Illinois Pollution Control Board, Section 35) and the federal USEPA (1986a) criteria.

Statistical Analysis of the Monitoring Data

The results of the Tier I analysis were presented in the preceding sections of this chapter. The same data bases were analyzed in Tier II, i.e., the MWRDGC and IEPA monitoring data. These data bases included an incomplete time series of samples taken infrequently. The log-normal plotting and analysis provided the probabilities of excursions of the Illinois General Use and Indigenous Aquatic Life standards. Note that these standards represent absolute minima of DO concentrations; therefore, using the probability of 99.8% of no excursion is only a statistical approximation. The probabilities of nonexceedance (compliance) of the standard (note that DO "nonexceedance" implies that the measured or statistically extrapolated concentrations are greater than or equal to the standard). An example of the statistical plotting of DO concentrations is shown on Figure 2.26. Complete statistical analyses and plots are in Appendix B. Table 2.15 contains the probabilities of excursions of the DO standards, both General Use and Indigenous Aquatic Life Use, are presented in Table 2.15. Dissolved oxygen measurements were collected as individual grab samples collected on a monthly or weekly basis. Analysis of compliance with the 16-hour duration criteria in the General Use Standard is, therefore, not possible from the available data.

Table 2.15 Probabilities of No Excursion of the 5 Mg/l DO Standard Obtained from Statistical Analyses

		General Use 5 mg/L	Indigenous Aq. Life 4 mg/L	
IEPA G-23		75 %	95%	Brandon Road Dam pool (Joliet)
MWRDGC	93	80 %	95%	Brandon Road Dam pool (Joliet)
MWRDGC	94	99 %	>99.8	Dresden Island Dam pool (Empress)
MWRDGC	95	>99.8 %	NA	Dresden Island Dam pool (I-55)

Knowing that the 1B3 probability of exceedance (once in 3 years) has approximately the same probability as occurrence of the 7Q10 low flow, the 99.8% or greater probability of no excursion would imply compliance with the standard. Thus, the General Use DO standard is not met in the Brandon pool. Note that the federal DO criteria are more lenient since they are related to the type of biota residing in the water body, presence or absence of the early life forms and the standards are compared to low 7 day, once in 10 years average concentrations (the higher value) and, maybe, to 24-hour averages rather than instantaneous means for the minimum DO. The statistical analysis documented that only DO concentrations near the I-55 reach were in compliance with the General Use standard and the MWRDGC sampling site 94 (Empress Casino) was also near compliance. With respect to the indigenous life Illinois standard of 4 mg/L, all sites located in the Dresden Island Pool were in compliance while none in the Brandon Road pool complied.

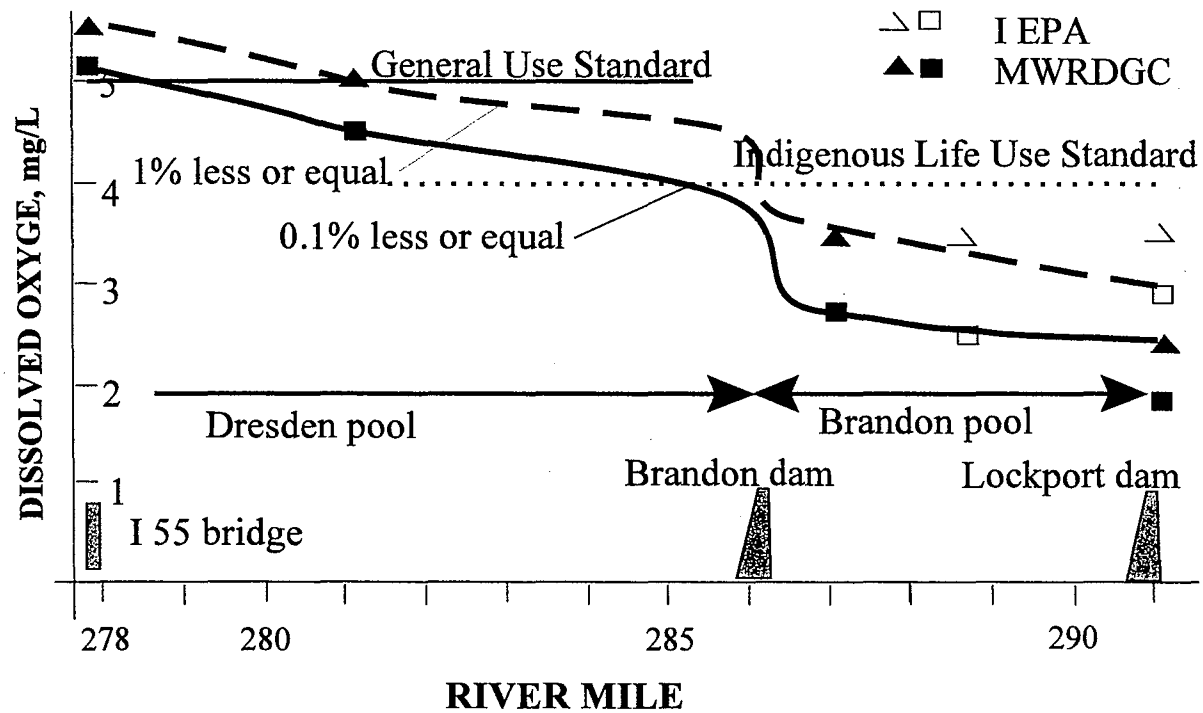


Figure 2.27 Longitudinal Plot of Do Concentrations Corresponding to 0.1 and 1 Percent Probability of Being less in the Lower Des Plaines River

The measured values were statistically extrapolated to 99.8 or 99 percentile values. The longitudinal plot of these statistical values is shown on Figure 2.27.

The plot of statistical values is not designed to detect the significant aeration effect of the Brandon Road Dam documented in Table 1.3 taken from the Butts et al. (1975) study. Butts et al. study documented that the river flow over the Brandon Road Dam can add as much as 5 mg/L of DO to the flow entering the Dresden Island Pool.

DO Concentrations at the Reference Sites

The rivers of the State of Illinois generally have a problem with meeting the General Use standard. This is shown on the log-normal plots of two reference streams; Figure 2.28 is the DO probability plot for the Kankakee River and 2.29 is the plot for the Green River. The Kankakee river is mostly free flowing, while the Green River is partially modified by channelization.

Assessment of reference streams is needed and useful for adjustment of the standard. The USEPA (1986) water quality criteria allows an adjustment of the standard:

> *Where natural conditions alone create dissolved oxygen concentrations less than 110% of the applicable criteria means or minima or both, the minimum acceptable concentrations is set at 90% of the natural concentration. ..... Absolutely no anthropogenic dissolved oxygen depression of the potentially lethal area below the 1-day minimum should be allowed unless special care is taken to ascertain the tolerance of resident species to low dissolved oxygen.*

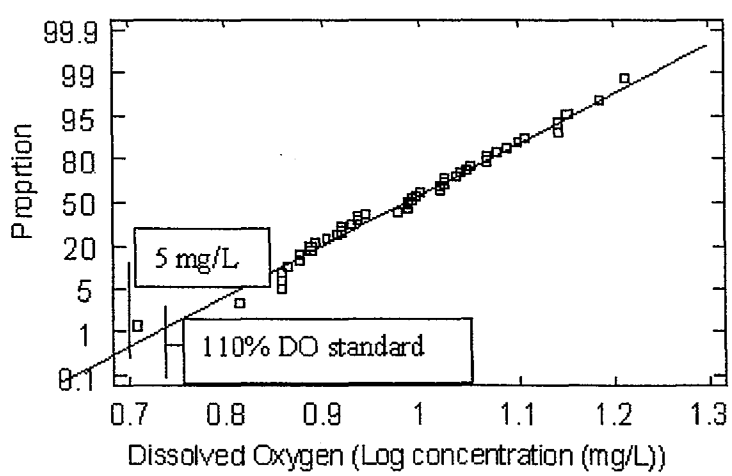


Figure 2.28 Measured DO Concentrations of the Kankakee River in Momence

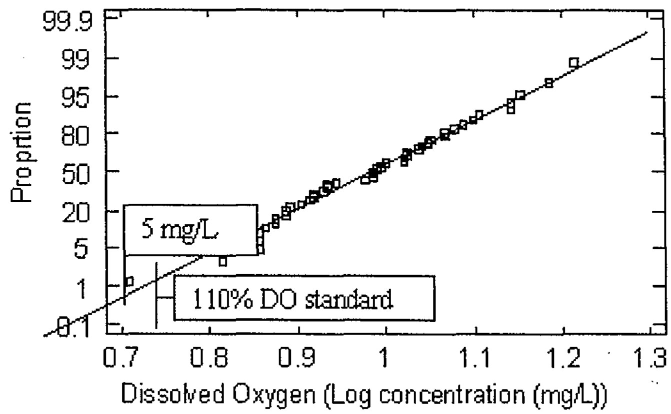


Figure 2.29 Probabilistic Plot of DO Concentrations of the Green River

Although none of the measured DO concentrations of the Kankakee and Green Rivers were below the 5 mg/L standard, at least one measurement at each river was below the 110% value. These measurements cannot be discounted as outliers because they fit the log-normal probability distribution. Furthermore, the 99.8 percentile values are about 4 mg/L. This indicates difficulties of meeting the 5 mg/L absolute minimum of the General Use DO standard even in the reference waters.

Continuous Monitoring by MWRDGC in Joliet and by Midwest Generation at I-55

The Metropolitan Water Reclamation District of Greater Chicago and Midwest Generation have installed continuing dissolved oxygen monitors. In Joliet at Jefferson Street (MWRDGC 93) the monitoring is operated by MWRDGC, and at I-55, by Midwest Generation. The continuous monitoring provides invaluable information on the course of DO concentrations. This is the only possible way to assess the short duration minima and hypothesize on the causes of the low DO and its duration. For example, DO fluctuates during the day as a result of algal activity in nutrient enriched streams, exhibiting the lowest summer DO concentrations in the late night and early morning hours and potential oversaturation in late afternoon. On cloudy days, algal respiration may bring dissolved oxygen to very low levels.

Figure 2.30 shows side-by-side DO concentrations in Joliet and I-55 in the summer of 2000. In this year, the I-55 site fully complied with the 5 mg/L standard. However, violations of the 4 mg/L Indigenous Life Use were measured in the Brandon Road pool by the MWRDGC Jefferson Street continuous monitoring station. Both sides exhibited significant daily fluctuations of DO caused by algal activity in the pools. The difference in average DO between the Brandon pool and I-55 was about 2 mg/L. On Figure 2.30, the General Use DO standard applicable to I-55 is 5 mg/L, that for the Brandon Pool (MWRDGC 93) is 4 mg/L.

Figure 2.31 is the plot of continuos DO monitoring at I-55 by Midwest Generation in July 1999.

During this year, DO concentrations dropped below 5 mg/L. One interesting conclusion is that, based on the grab sampling program and the minimum DO standard of 5 mg/L, the Illinois General Use is nearly met at the I-55 bridge. From the continuous monitoring program conducted by Midwest Generation, one could arrive at almost the same conclusion. The days at which the DO in the continuous monitoring program dropped below the 5 mg/L limit in the 1997 -2000 period were:

Date	Maximum excursion below 5 mg/L	Duration Hrs	Major cause
August 4, 97	0.25 mg/L*	14	Unknown
June 27, 1998	0.3 mg/L*	3	Unknown
July 11, 1999	2.1 mg/L	10	Mostly algal respiration
July 12, 1999	2.1 mg/L	3	Mostly algal respiration
July 27, 1999	0.3 mg/L*	2	Algal respiration
August 16, 1999	1.1 mg/L	8	Algal respiration
No excursions in 2000			

* within the measurement error

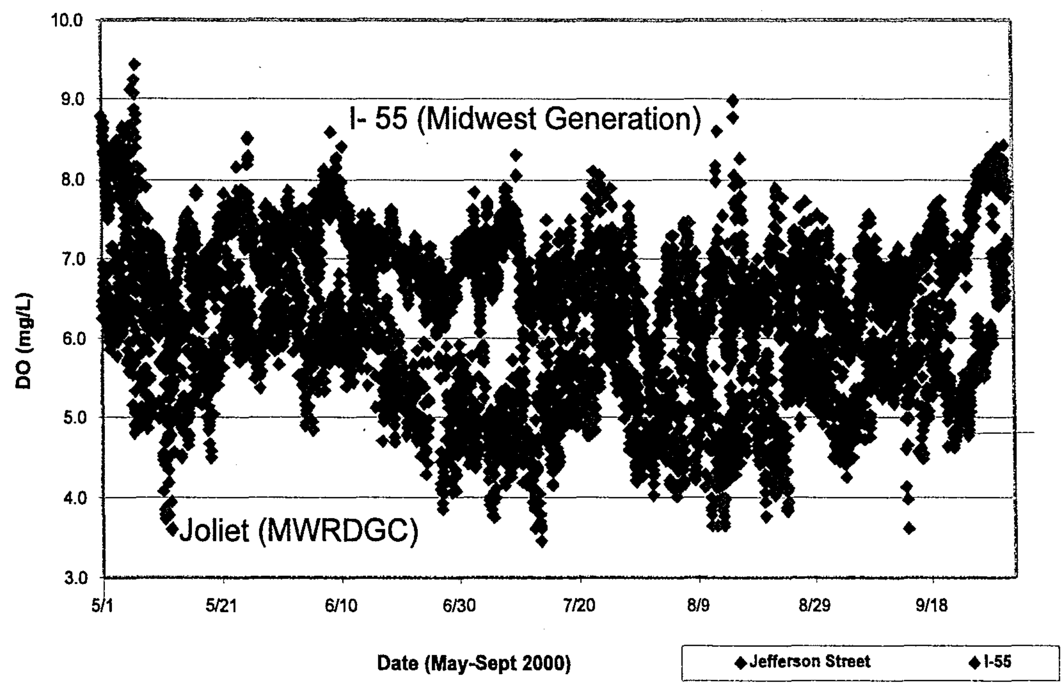


Figure 2.30 Continuous Side by Side DO Monitoring in Joliet (MWRD 93) And I-55 (Midwest Generation) During Summer of 2000. Significant Daily DO Fluctuations Were Recorded.

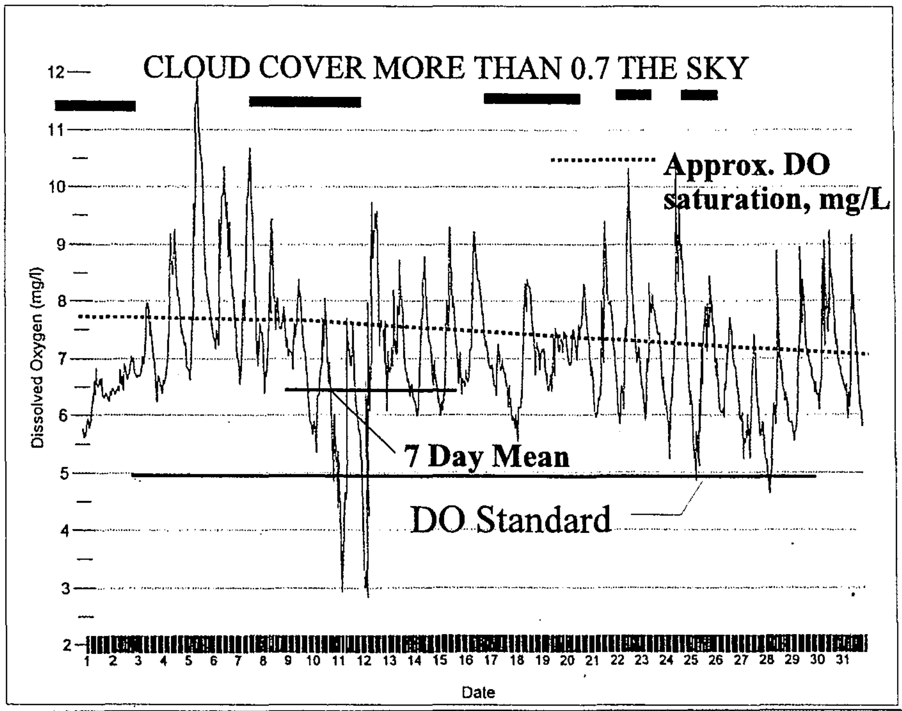


Figure 2.31 Continuous DO Monitoring by Midwest Generation at I-55 in July 1999

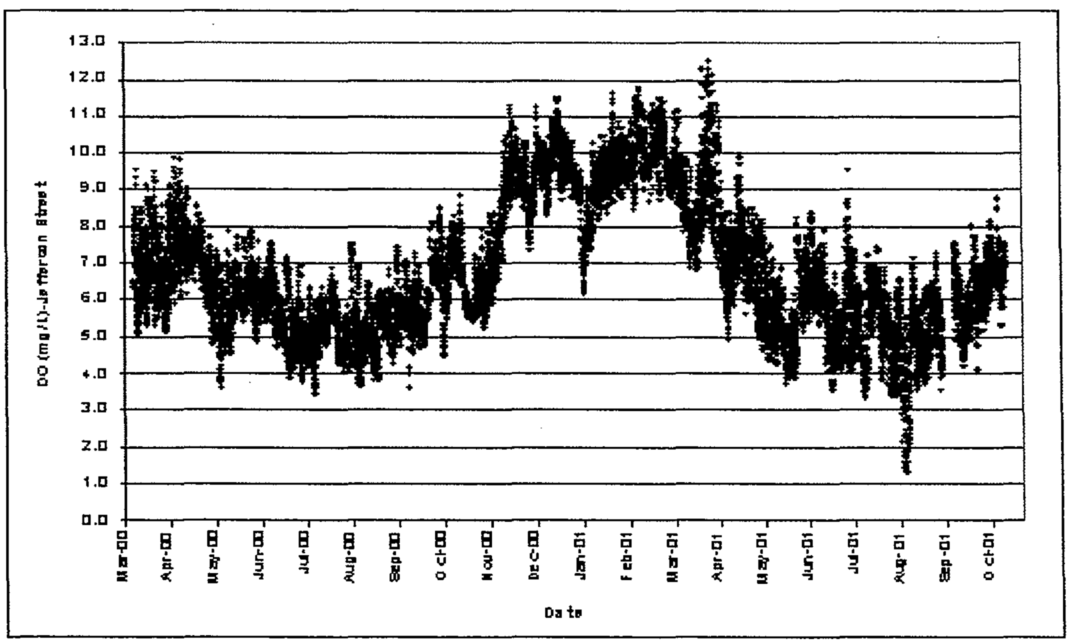


Figure 2.32 Continuous DO Concentrations at MWRDGC 93 (Jefferson Street in Joliet) of the Brandon Road Pool for the 2000-2001 Period. The Monitoring Station Is Operated by the Metropolitan Water Reclamation District of Greater Chicago.

Hence, there were only 2 significant excursions from the 5 mg/L General Use Standard at I-55 in the 4-year period of 1997-2000, or 99.8 % of measurements were apparently above the 5 mg/L minimum standard. The remaining three short duration excursions during the night or early morning hours, caused most likely by algal respiration, were within the measuring error of the DO monitor. In view of the comment in the federal criteria document (USEPA, 1986), they could be either disregarded or given less weight. The statistical extreme value results of the continuous monitoring are very close to the log-normal probability projection of grab sampling at this location (MWRDGC 95) by the Metropolitan Water Reclamation District of Greater Chicago. The period of the lowest DO concentrations at the I-55 bridge is shown on Figure 2.31. On July 11 and 12, 1999, hourly minima were around 3 mg/L. Because the flow was not less than the 7Q10, these excursions would represent a violation of the Illinois standard. However, upon closer investigation of the pattern, it was noted that the average DO during the 7-day period of low concentration was more than 6 mg/L and the minimum average daily concentrations were not far above 5 mg/L. Hence, the federal DO criterion was not violated.

At the MWRDGC Station 93 in Joliet (Figure 2.32) there were several incidences of the DO being below the secondary standard of 4 mg/L during the period of 2000 - 2001 and one incidence of the DO dropping to about 1.0 mg/L (August 2001), which is lethal to fish.

One problem that became apparent upon analyzing the continuos monitoring was that the DO concentrations at both stations (I-55 and Joliet) were greatly affected by photosynthesis and respiration caused by very high nutrient levels. Daily DO fluctuations by as much as 3 - 4 mg/L (low in the early morning hours and high in the late afternoon) are common during summer months at the I-55 bridge (Figures 2.30 and 2.31) and at the MWRDGC 93 in Joliet in the Brandon Pool (Figures 2.30 and 2.32). Oversaturation with DO exceeding 150% saturation values were measured in July and August 1999. During early morning hours, on the other hand, DO concentrations at some instances dropped to very low values because algal respiration during night hours created a sink of oxygen. Algal respiration is also a problem during cloudy days. A question may be asked whether the very low DO in July 1999 was caused by deoxygenation of BOD or by algal respiration on cloudy days.

Occurrences of the significant daily DO fluctuations during summer months, DO oversaturation and high nutrients level are obviously signs and symptoms of eutrophication. They are also signs that the decomposition of organic biodegradable carbonaceous pollution from wastewater effluents by heterotrophic bacteria has been mostly completed. This is due to the fact that the heterotrophic bacteria decomposing the BOD have a greater growth rate than the autotrophic algae. If organic BOD type pollution from waste water effluents had been present at higher concentrations, heterotrophs would have decomposed both algal biomass and BOD. Algal biomass does not develop in waters that have high organic biodegradable pollution. However, this does not imply that high algal densities are preferable. On days with not enough sun light and during night hours algae respire or, after die-off, are decomposed by the heterotrophic bacteria (decomposers) and exert a high DO demand both in water and in the sediment (sediment oxygen demand). This may explain the large DO drop at I-55 in July 1999 shown on Figure 2.31.

To prove that the large excursion on July 10-12, 1999 at I-55 was caused by algal respiration and not by distant wastewater discharges, we plotted occurrences of cloudy days on Figure 2.31. As stated before, the significant daily variations of the dissolved oxygen content are caused by nutrient enrichment that stimulates excessive algal growth. During sunlight, algae produce oxygen that is manifested by high oxygen concentrations, exceeding saturation (plotted on Figure 2.31 as a dotted line). Supersaturation of the water with DO (i.e., DO is greater than the saturation concentration) can be achieved only by algal photosynthetic activity. During night hours or during days with full or near full cloud cover, algae do not produce oxygen, they use DO by their respiration. Also, the excess oxygen (over saturation limit) is lost from the stream by deaeration, which is the opposite of reaeration through which the oxygen is exchanged through the water-air interface. Figure 2.31 does show that the periods of cloudy days coincide with the period of lower DO concentrations. *Thus, we have concluded that the DO drop on July 10-12, 1999 might have been caused by algal respiration due to a lack of light energy input. However, this is not completely a natural phenomenon, it is a water quality problem that could be alleviated by control of nutrient levels in the river and an eventual TMDL designed to address such problem should be focused mainly on the nutrient levels.*

The federal USEPA (1986) water quality criteria provide a partial remedy to this problem. The USEPA criteria document states that during periodic cycles of dissolved oxygen concentrations, minima lower than acceptable constant exposure are tolerable so long as:

- *the average properly calculated concentration attained meets or exceeds the criterion;*
- *the minima are not unduly stressful and clearly are not lethal.*

This wording allows to consider daily mean instead of instantaneous minimum for waters that are affected by photosynthetic oxygen production and algal respiration. This contradicts the wording of the DO criterion in Table 2.1. However, this absolute minimum should not be below 3 mg/L, which is the lethal threshold for fish. There has been a considerable and unresolved discussion among the USEPA water quality standards specialist as to whether the daily minimum DO concentration is to be applied to an instantaneous minimum or lowest mean daily concentration[7]. It should be noted that even on July 10-12,1999 with the lowest DO minima, the average 24-hr DO concentrations were above the 5 mg/L standard. As pointed out above, *because these short term excursions occur with a frequency of less than once in three years, they may not constitute a violation of federal criteria. However, they do violate the current "no excursions except during 7Q10" Illinois DO standard for the General Use.*

Relation of the DO Concentrations to Flow

To investigate whether the DO excursions are dry weather or wet weather problem, DO concentrations at the MWRDGC 93 (Joliet) were plotted vs. flow (Figure 2.33). If the cause of the low DO problem was an upstream point source, the lowest DO concentrations would coincide with the lowest flow and the efforts should be focused on the reductions of BOD and NOD (nitrogenous

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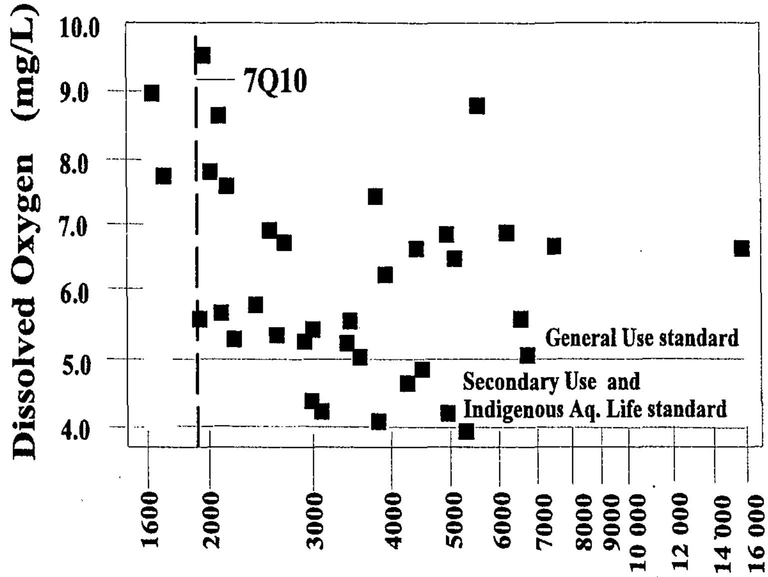
[7] Personal communication by Charles Delos (USEPA) to Vladimir Novotny



FLOW (cfs)

Figure 2.33 Plot of DO Concentration at the MWRDGC Site 93 in Joliet on Brandon Pool vs. flow.

oxygen demand) inputs from upstream sources (i.e., the MWRDGC effluent discharges into the Chicago waterways). The plot reveals that the lowest DOs occur at medium flows while DO at flow approaching the 7Q10 minimum flow do not cause excursions. This would indicate that upstream discharges of wet weather overflows (remaining CSOs) or, possibly algal respiration, may be a cause. Therefore, it is expected that further reduction of CSOs by fully implementing the TARP project will have a beneficial effect on the DO concentrations in the Brandon pool.

Attainability of the DO Standard

<u>Historic Comparison</u>

Upon comparing the historic DO concentrations measured by Butts et al. (1975) with the most recent DO measurements, one cannot help to notice the tremendous progress in wastewater treatment achieved by the MWRDGC and other dischargers on the Des Plaines River. Figure 2.34 compares the DO levels in Brandon Pool in Joliet and I-55 in 1972 and 2000.

In 1972, the DO concentrations could not meet an interim standard of 2 mg/L (Butts et al., 1975) applied at that time. In 2000, maintaining average daily DO of 5 mg/L in Brandon pool is a common occurrence, although on occasion the minimal DO may drop below 3 mg/L. Thus, the largest improvement occurred in the Brandon pool. The Dresden Island pool (I-55) concentrations generally met the 5 mg/L standard in 1972 due to, as pointed out previously, the high aeration capacity of the Brandon Pool Dam. However, Butts et al. pointed out to the unrealistic situation of applying the dual standard to the Dresden Island Pool. In 1972, the DO concentrations in the Dresden pool were higher upstream of I-55 (Secondary Contact and Indigenous Life standard) than those downstream of I-55(General Use). In 1972, there was no correlation between the magnitude of flow and the minimum DO concentrations. The minimum DO concentrations for high flow sampling days were in the same range as those for low flow conditions. It was explained that during high flows, the oxygen consuming loads from CSOs and urban runoff increase in proportion to the increase flow. Further improvements in Dresden Island Pool by stream aeration (both natural or human induced such as dam aeration or side stream aeration) may be difficult due to the high temperature in the Upper Dresden pool caused by the heated discharges from the Midwest Generation plants. This is because the maximum aeration rate is proportional to the oxygen deficit expressed as

$$r_{O2} = k(C_s - C)$$

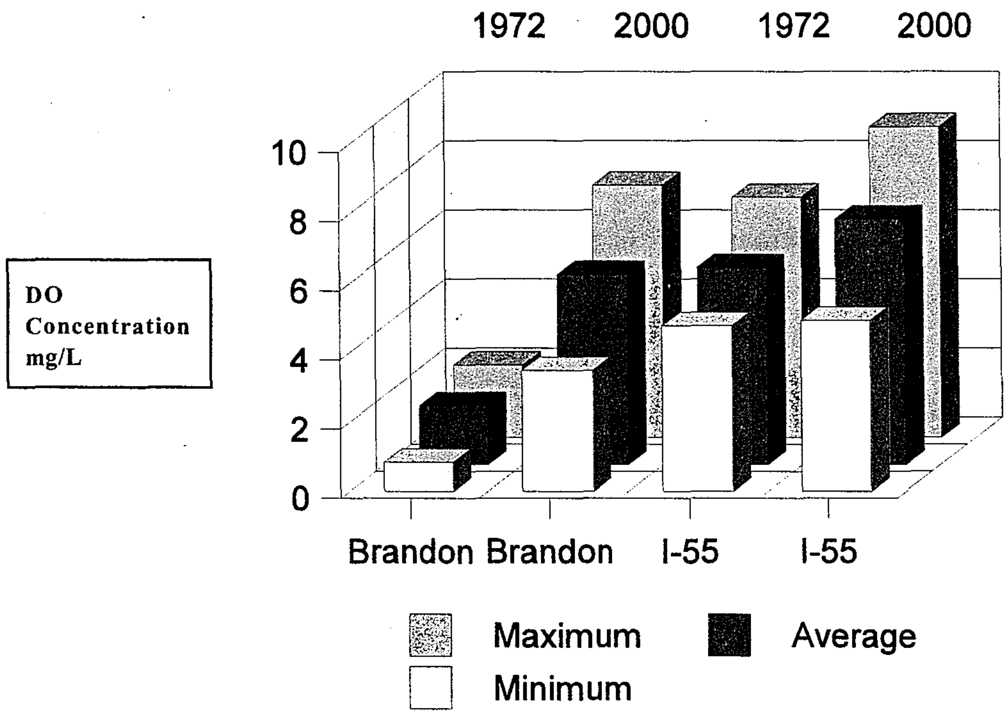


Figure 2.34 Changes in Dissolved Oxygen Concentrations from 1972 to 2000

where C_s is the saturation concentration related to temperature and C is the DO concentration of water. The maximum temperatures in the upper part of the Dresden Island pool during summer reach 35 to 37°C (100°F) (Wozniak, 2002) during which the oxygen saturation concentration is smaller. At 37°C the oxygen saturation is

$$C_s = 14.652 - 0.41022x38 + 0.007991x38^2 - 0.000077774x38^3 = 6.3\,mg\,/\,L$$

For polluted water the DO saturation would be less, possibly less than 6 mg/L. Theoretically, DO concentrations, in absence of photosynthesis, cannot reach or exceed the saturation values. There are many literature sources that explain the phenomenon of reaeration of the receiving water bodies. One of the latest ones is Chapra (1997). Also Butts et al. (1975) includes a very good discussion on the weir and in-stream aeration.

For comparison, the summer high temperatures upstream of the Midwest Generation outfalls are about 6°C (10°F) less or about 32°C (which is the General Use standard for I-55). At this temperature, the oxygen saturation is C_s = 7.15 mg/L, or 7 mg/L for polluted water.

This calculation of the oxygen saturation indicates that, due to the high temperatures, attainment of the 6 mg/L DO concentrations in the Upper Dresden pool under present thermal loads from the Midwest Generation plants is impossible solely by aeration of the flow. Actually, oxygen in excess of 6 mg/L delivered by photosynthesis and aeration of the Brandon Pool dam during lower temperatures upstream of the power plants is being lost from the river due to the higher temperature.

DO Modeling

Classical DO modeling may assist in understanding the processes. The dissolved oxygen in a stream is affected by a number of processes that were summarized by Thomann and Mueller (1987 and also by Novotny (2002) as:

- Sinks of oxygen, that is the biochemical and biological processes that use oxygen, include:
  1. Deoxygenation of biodegradable organics whereby bacteria and fungi (decomposers) utilize oxygen in the biooxidation-decomposition process.
  2. Sediment oxygen demand (SOD), where oxygen is utilized by the upper layers of the bottom sediment deposits.
  3. Nitrification, in which oxygen is utilized during oxidation of ammonia and organic nitrogen to nitrates.
  4. Respiration by algae and aquatic vascular plants which use oxygen during night hours or during heavy cloud overcast to sustain their living processes.
  5. DO from an oversaturated stream and during high temperatures can also be lost by deaeration which is a reverse process of reaeration.

- Oxygen sources are:
  1. Atmospheric reaeration, where oxygen is transported from the air into the water turbulence at the water interface or can be supplied by flow turbulence at dams, in-stream or side stream aeration, or turbine aeration.
  2. Photosynthesis, where chlorophyll-containing organisms (producers such as algae and aquatic plants) convert CO_2 (or alkalinity of water) to organic matter with a consequent production of oxygen on days with minimal cloud cover. Photosynthesis is driven by the nutrients and light energy.

The rate of each process and reaction is a function of temperature. All processes mentioned above are present in the Lower Des Plaines River. Therefore, no single cause of the low DO can be pinpointed. Some oxygen sinks are controllable (e.g., reduction of BOD and ammonium discharges from effluents and CSOs), some are less controllable (e.g., the effect on nutrients and temperature). Some are uncontrollable (e.g., sunlight that drives photosynthesis or cloudiness that works in the other direction, or sediment oxygen demand of sediments in slow velocity depositional reaches).

The oxygen balance and reactions resulting in variations of DO concentrations in a complex system such as the Lower Des Plaines River can be best studied by a water quality model. Several water quality/DO models have been developed in the past. Butts et al. (1975) developed a classic steady state DO model that considered dissolved oxygen consumed biologically within a reach by deoxygenation, nitrification, and sediment oxygen demand. From the reasons stated above, i.e., presence of a high organic pollution content that suppressed algal development and photosynthesis, Butts et al. did not include algal photosynthesis and respiration in the model.

The classic dissolved oxygen model that would be applicable to the Des Plaines River begins with the differential equation describing the oxygen mass balance (Thoman and Mueller, 1987; Krenkel and Novotny, 1980)

$$\frac{dC}{dt} = K_a D - K_d L_r - K_N L_N - \frac{S_B}{H} + (P - R)$$

where

C = concentration of dissolved oxygen, mg/L
$D = (C_s - C)$ = oxygen deficit, mg/L
C_s = oxygen saturation, mg/L
L_r = concentration of carbonaceous biochemical oxygen demand - BOD, mg/L
L_N = concentration of nitrogenous oxygen demand - NOD, mg/L
S_B = benthic oxygen demand, g/m^2 - day
P = photosynthetic oxygen gain, mg/L-day
R = oxygen loss by algal respiration, mg/L-day
K_a = coefficient of reaeration, day^{-1}
K_d = coefficient of deoxygenation, day^{-1}
K_N = coefficient of nitrification, day^{-1}
H = depth of the reach, meters

Variables C_s, K_a, K_d, K_N, S_B, P, and R are temperature dependent. The temperature effect on the reaction rates and benthic oxygen demand are described by

$$K_T = K_{20}\,\theta^{(T-20)}$$

where T = temperature in °C
θ = thermal factor, which has the following accepted values

Deoxygenation rate coefficient	$\theta = 1.047$
Reaeration rate coefficient	$\theta = 1.024$

Nitrification rate coefficient $\theta = 1.1$
Sediment oxygen demand $\theta = 1.05 - 1.06$

The oxygen mass balance equation has to be coupled with equations describing the removal of BOD and NOD by biochemical deoxygenation and nitrification processes. The saturation DO concentration is calculated by the equation given in the preceding section.

Photosynthesis and respiration is related to the nutrient level (nitrogen and phosphorus) that stimulates algal growth. The product, besides the DO effect, is the concentration of chlorophyll-a. These equations form a basis for most water quality models, including QUAL2E.

A QUAL2E model was developed for the Lower Des Plaines River to assess the attainability of the General Use (Dresden Island pool) and Indigenous Aquatic Life Use in the Brandon pool by the Institute for Urban Environmental Risk Management of Marquette University (Appendix D) under the direction of Dr. Charles Melching. The model and the model parameters were provided by the Metropolitan Water Reclamation District of Greater Chicago. The courtesy of providing the model is greatly appreciated. The original model was developed for MWRDGC by Camp, Dresser and McKee (1992). The courtesy of providing the model does not imply an endorsement of the results by MWRDGC. The results and interpretations are those of the consultant (AquaNova International/Hey and Associates) and not of MWRDGC.

The basic question and task addressed by the model were whether the standards of 4 mg/L in Brandon Pool and 5 mg/L (6 mg/L) in the entire Dresden Island pool are achievable, provided that minimum DO of 4 mg/L is achieved at the Lockport Lock and Dam site (MWRDGC. 92). Presently, the DO concentrations drop occasionally below the 4 mg/L at both Lockport and Joliet sites.

The second question is what measures should be taken to increase the DO concentrations to meet these standards. Table 1.3 shows that as far back as in the 1970s, the 5 mg/L dissolved oxygen concentrations of 5 mg/L were routinely measured downstream in the Upper Dresden Island pool in the tailwater of the Brandon Road Dam, owing to the high aeration capability of the dam (Figure 1.3). The aeration capability of dams is proportional to the upstream oxygen deficit that in the 1970s was large because the DO concentrations upstream of the dam were commonly around 1 mg/L.

Currently, the DO concentration in the Brandon Pool are mostly above 4 mg/L.

QUAL2E Modeling Results

The QUAL2E report by the Institute for Urban Environmental Risk Management of Marquette University is included in Appendix D. The model assumes that the major transport mechanisms for chemical constituents are advection, and dispersion, and that these mechanisms are significant only along the main direction of flow. It allows for multiple waste discharges, withdrawals, tributaries flows, and incremental inflow and outflow.

Hydraulically, QUAL2E is limited to the simulation of the time periods during which both the stream flow in riverbasins and input waste loads are essentially constant. QUAL2E can operate either as a steady state or as a quasi-dynamic model. When simulated as a steady state model, it can be used to study the impact of waste loads on stream water quality and also can be used in conjunction with a field sampling program to identify the magnitude and quality characteristics of nonpoint source waste loads. By operating the model dynamically, the user can study the effects of diurnal variations of algal photosynthesis on water quality.

The application of the QUAL2E model to the study area requires several assumptions to be made. Hydrologically, QUAL2E is limited to the simulation of the periods during which both the river flow and plant flows (water reclamation plants, and tributaries) are constant. Rivers must also be well mixed horizontally and vertically, and the major transportation mechanisms, advection and dispersion, are significant only along the main direction of flow. The data presented in this report will indicate that these assumptions are upheld for the application of the model.

The model has 6 reaches with a computational element length of 0.25 mile. It begins at the Lockport Lock and Dam and ends at the I-55 bridge, a distance of 13.25 miles. The reaches and the elements of the model is given in Table 2.16. A schematic diagram of the reaches and location of the point sources are given in Figure 2.35.

Table 2.16 Model Reaches and Elements

Reach #	Starting Point (River Mile)	Ending Point (River Mile)	Number of Elements	Location
1	291	290	4	Downstream of LP & D - CSSC*
2	290	287.25	11	Brandon Pool- D. P. R**
3	287.25	286	5	Brandon Pool- D.P.R
4	286	285.25	3	Dresden Pool –D.P.R
5	285.25	280.25	20	Dresden Pool –D.P.R
6	280.25	277.75	10	Dresden Pool –D.P.R

*CSSC = Chicago Sanitary Ship Canal; **D.P.R = Des Plaines River

The channel cross section data were obtained to determine approximate channel dimension. A total of 40 cross sections were provided. Based on the cross section data, trapezoidal approximation was done to obtain average bottom and top width, and water depth to use in the QUAL2E model. Reach slopes were calculated using bottom elevations at different points in the reach. Table 2.17 shows a summary of the cross section data developed for the model reaches.

The model simulates effects of carbonaceous BOD removal, nitrification, sediment oxygen demand, photosynthesis and respiration, and aeration effects of the Brandon Road Dam. Water quality data for calibration and verification processes were taken from the CDM (1992) report. Calibration/ verification sampling was performed in September/October 1990, May/June 1991, and July/August 1991. Each sampling event consisted of six passes. Each pass was 8 hours long for a total of 48 hours os sampling.

Table 2.17. Summary of the Cross Section Data

Reach #	Bottom width (ft)	Top Width (ft)	Channel slope (ft/ft)	Side slope 1 (H/V) (ft/ft)	Side slope 2 (H/V) (ft/ft)
1	230	310	0.00090	1.50	1.50
2	275	350	0.00090	0.60	0.70
3	1100	1100	0.00180	0.001	0.001
4	1000	1000	0.00110	0.001	0.001
5	200	1000	0.00023	17.50	8.5
6	280	1000	0.00029	9.00	10

Flow monitoring was performed at each significant hydrologic input into the modeled section of the river that included CSSC at Lockport, Des Plaines River, Joliet East and West, and Mobil WWTP. The flow from the CSSC constituted from 77 to 95 % of the total flow of the river. The flow ranged from 3000 to 4000 cfs.

Aeration of the river. The reaeration coefficient calculated by the O'Connors and Dobbins formula (see Krenkel and Novotny [1980] for the description of the model and formula) was 0.15 day^{-1}. This reaeration rate was verified by the calibration and verification runs. The dam aeration is calculated by the formula developed from weir aeration observations in England by the Water Pollution Research Laboratory (1973)

$$\frac{D_u}{D_d} = 1 + 0.38\, a\, b\, h(1 - 0.11\, h)(1 + 0.046\, T)$$

where D_u and D_d are respective upstream oxygen and downstream deficits of oxygen ($D = C_s - C$), C is the dissolved oxygen concentration, C_s is the saturation DO concentration, a and b are coefficients, h is the height of the dam in meters and T is temperature. The factors used in the model were a=1.1 and b = 1.8. The height of the dam is 10.3 meters (34 feet). The reaeration rate of the Brandon Dam is significant and, essentially, the DO immediately downstream of the dam should be near the saturation concentration. However, at such high aeration rates, some of the oxygen is not immediately dissolved and can be lost. The difference can be as much as 1 mg/L.

Figure 2.36 presents typical DO calculations by the QUAL2 model. Additional calibration, verification and production runs are in Appendix D. The data and calculations document that:

- The BOD_5 in both pools is relatively low and steady, meaning that the BOD in the pool is not being degraded, essentially it is a residual BOD that could also be related to algal production of the dissolved organic carbon.

- Algae growth in the modeled runs was mostly in the Dresden Island pool. In 1991, the Chlorophyll-a content was less than 10 µg/L.

- If the DO in the Brandon pool is about 4 mg/L, aeration at the Brandon Road Dam can bring the DO content close to the saturation value and result in the initial oxygen concentration in the Dresden Island Pool of greater than 5 mg/L.

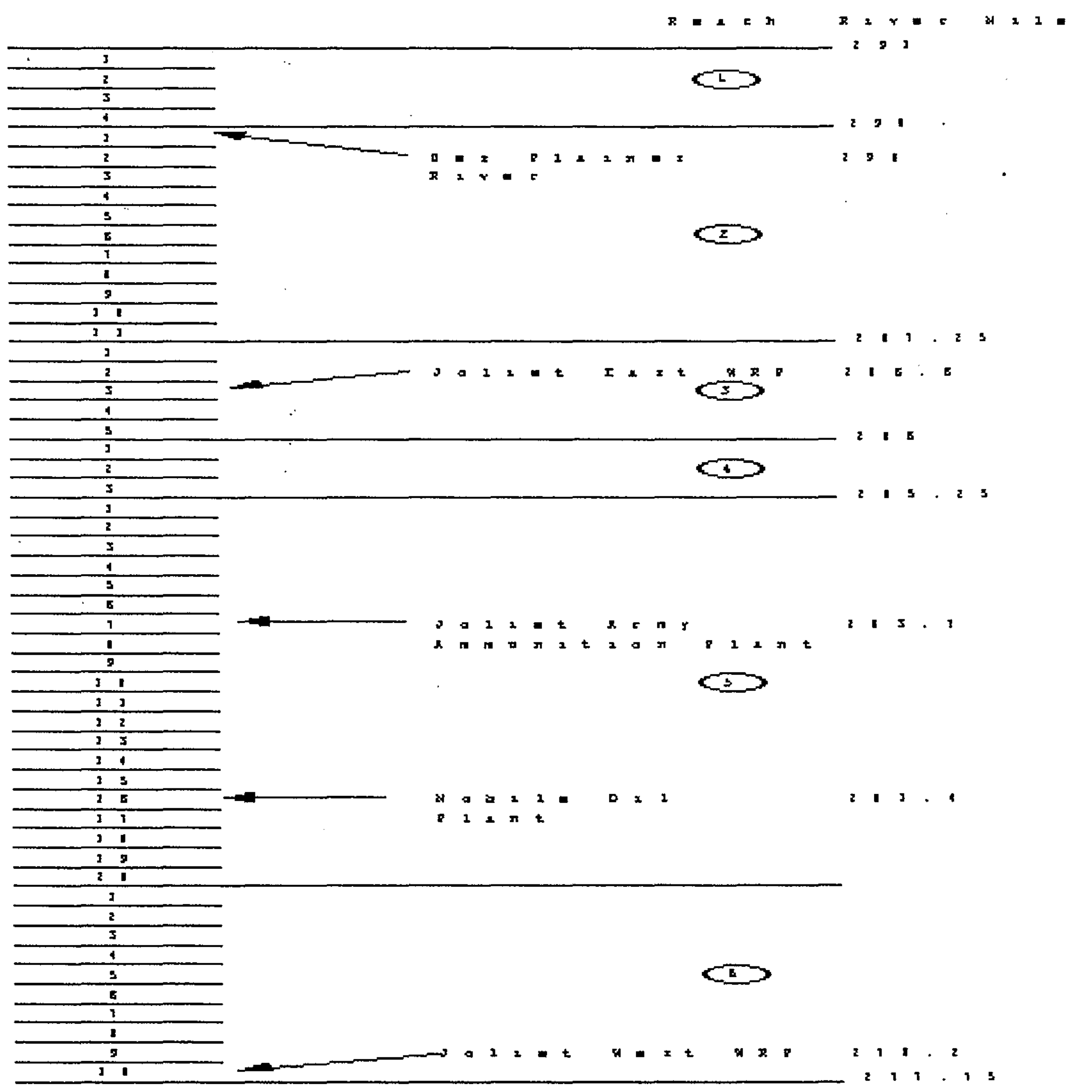


Figure 2.35. Schematic diagram of the Lower Des Plaines River QUAL2E Model

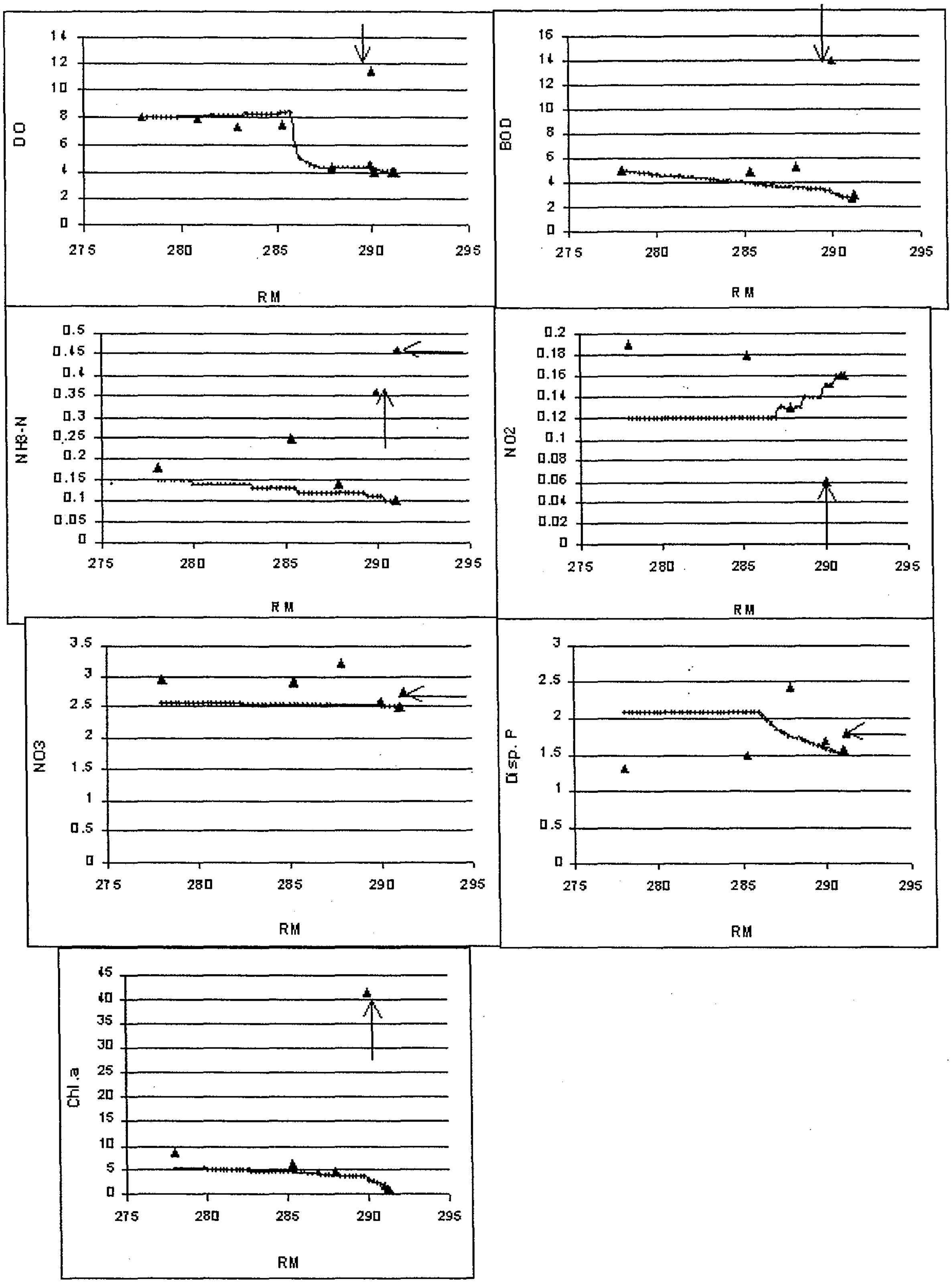


Figure 2.36 QUAL2E Results for July 1991 Verification Runs

UAA Six Reasons Issues

(1) Naturally occurring pollutant concentrations prevent the attainment of the use

At the two selected reference streams (the Kankakee and Green Rivers), the General Use standard of 5 mg/L was met at the 99% compliance level. No measured concentrations were below the standard; however, at each site one measurement was 5 mg/L. Statistically, this means that there could be a 1% probability that measured DO levels could drop below the General Use standard. Federal USEPA (1986) DO criteria state that if the concentrations at the reference sites drop below 110% of the criterion (i.e., 5.5 mg/L), states can establish a DO standard at 90% of the reference value, i.e., 4.5 mg/L.

The instantaneous DO excursions in the Dresden Island pool were caused by nutrient enrichment and lack of light on cloudy summer days. While nutrient enrichment is due to discharges of nitrogen in the upstream effluents and, to a lesser dégree from nonpoint source, the reduction of the light energy on cloudy days is a natural cause of decreasing DO concentrations. The federal USEPA suggest to remedy the problem by allowing average daily concentrations to be considered in situations where significant daily fluctuation of DO occur.

(2) Natural, intermittent (ephemeral) or low flow or water levels prevent the attainment of the use unless these conditions may be compensated for by the discharge of a sufficient volume of effluent discharge without violating state conservation requirements.

This reason does not apply.

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place.

The DO problem in the Brandon Road Dam pool can be corrected, e.g., by providing more aeration at the Lockport dam. One possibility is to use turbines that are designed to aerate or allowing some flow to pass over the spillway (currently rarely used) either by gravity or by pumping. Side - stream aeration may not be possible due to a lack of available space. As it will be documented in subsequent chapters of this document, physical features of the Brandon Road Dam pool prevent development and propagation of early life forms. Consequently, following the USEPA (1986) criteria document, less stringent and attainable DO standard will be proposed for the Brandon Road Dam pool.

(4) Dams, diversions or other hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original conditions or to operate such modifications in a way that would result in the attainment of the use.

Although both Brandon Road Lock and Dam and Dresden Island pools represent highly modified water bodies, the General Use standard (applied to 24 hour average concentrations) is being attained and is attainable in the Dresden pool. A modified use and an attainable standard will be proposed for the Brandon Road Lock and Dam pool.

(5) ***Physical conditions associated with the natural features of the water body, such as the lack of proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to quality preclude attainment of aquatic life protection uses.***

This reason refers mainly to attainment of the use by developing a balance biota. It may not be applicable to the attainment of the dissolved oxygen standard.

(6) ***More stringent controls than those required by Sections 301(b) and 306 of the CWA would result in substantial and widespread adverse social and economic impact.***

The modeling study by QUAL2E model documented that if the DO concentration downstream of Lockport is at 4 mg/L or greater, minimum 4 mg/L DO concentrations can be maintained throughout the Brandon pool. The socio-economic issues of attaining the required DO concentration upstream of Lockport will be addressed in a subsequent UAA study of the Chicago Waterway System.

Conclusions on the DO Analysis

Brandon Road Dam Pool

- The dissolved oxygen concentrations in the Brandon pool are today significantly greater than those measured in the 1970s. Both grab samples and continuous monitoring show that in most times the Secondary Contact and Indigenous Aquatic Life standard of 4 mg/L is being met; however, the frequency of excursions at flows greater than those of 7Q10 and outside of the 7Q10 flows is not acceptable and, legally, these excursions would represent a violation of the Illinois rule of maintaining the standard at all times.

- Aeration by the release of the CSSC waters through the Lockport power house is not sufficient to guarantee the standard being met "at all times." If at least a part of the flow is aerated by allowing the flow discharge over the (unused) spillway of the Lockport Dam or by practicing turbine aeration it would be possible to meet the Illinois Secondary Use and Indigenous Aquatic Life Use standard.

- Because almost all BOD from upstream treatment plants has been removed by the treatment process at the upstream MWRDGC water reclamation plants and by self-purification during the time of travel between the MWRDGC plant effluents and Lockport, it is unlikely that further BOD removal at the MWRDGC plants would have an effect on the dissolved oxygen concentrations in the Brandon pool.

- Without significant aeration at the Lockport Dam or powerhouse, meeting the Illinois General use standard of 5 mg/L may not be attainable without a concurrent increase of the DO concentrations in the CSSC upstream of the Lockport dam to or above 5 mg/L. The appropriateness of the General Use standard for the Brandon pool will be extensively deliberated in Chapter 7. This Chapter will develop and propose a special use classification for this pool.

Dresden Island Pool

- Most of the time, dissolved oxygen concentrations in the Dresden Island Pool between the Brandon Road Dam and I-55 bridge meet the Illinois General Use standard of 5 mg/L. However, few excursions recorded by the continuous monitoring by Midwest Generation at the I-55 bridge violate the Illinois Water Quality Standard Rule of no excursions at all times.

- Meeting the 6 mg/L General Use standard in the Dresden pool for the minimum 16 hours is difficult in the Dresden pool during summer when the temperature of the pool is high.

- The saturation value is related to the temperature. Consequently, by increasing the temperature by heated discharges, part of DO gain at the Brandon Road Dam spillway may be lost. Because the saturation DO value at the 37°C (100°F) temperature is about 6 mg/L, meeting the 6 mg/L limit may not be possible during times when the temperature in the pool is near the standing Secondary Use and Indigenous Aquatic Life temperature maximum standard of 100°F.

- Aeration by the flow over the Brandon Road Dam brings the DO downstream of the dam close to the saturation value.

- The Dresden Island pool is eutrophic, which is exhibited by large diurnal DO variations during summer months and high nutrient concentrations. As a result, on occasions due to algal respiration, the minimum daily DO concentrations drop below 5 mg/L.

- While the Illinois General Use standard of 5 mg/L is being met at I-55, this standard may not be attainable in the Upper Dresden Island pool between the I-55 bridge and the Brandon Road Dam. The federal DO criterion of 5 mg/L for warm water bodies as formulated in the USEPA (1986) criteria document may be attainable, provided that the criterion frequency component of allowable excursions is considered and included into the Illinois General use standard. A proposal for a modification of the General Use standard for the Dresden pool is included in Chapter 8.

Temperature

Temperature is a major factor affecting the biological integrity of a water body. Excess temperature can affect chemical and biological reactions, decrease dissolved oxygen solubility, increases toxicity of ammonia, and affect metabolism of aquatic organisms.

The Lower Des Plaines River receives and carries thermal loads from three power plants located on the upstream CSSC (Will County) and on the investigated reach of the river (see Table 1.2). The Des Plaines River upstream of Lockport is a warmwater stream. However, its flow constitutes less than 10% of flow of the river in the Brandon Pool. Most of the flow comes from the Chicago Sanitary and Ship Canal.

The Lower Des Plaines River is loaded by heated discharges, primarily from power plants. The thermal loads and condenser temperature increases were listed in Table 1.2. The plants include two units in Joliet located on the upper portion of the Dresden Island pool (Figure 2.37) and Fisk, Crawford and Will County plants located upstream of the investigated reach (upstream of Lockport). The Joliet plants are located approximately 7.3 miles upstream of the I-55 bridge in the segment of the Des Plaines River designated as the Secondary Contact and Indigenous Aquatic Life use.

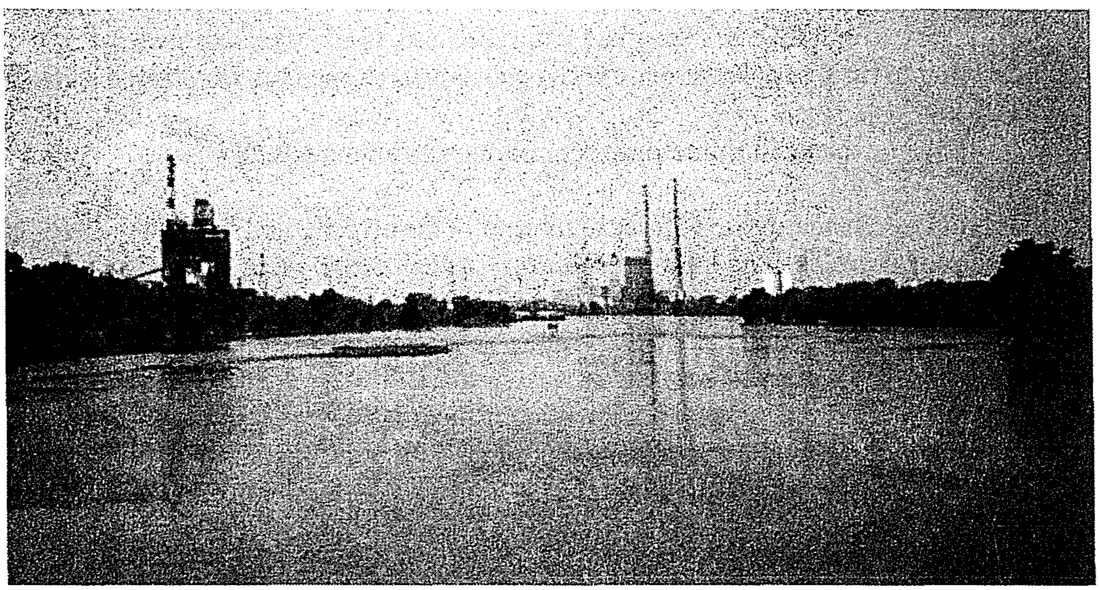


Figure 2.37 Two Thermal Power Plant Units Operated by the Midwest Generation in Joliet Located on the Upper Dresden Island Pool

The effluents of the MWRDGC treatment plants located upstream do not increase temperature during warm weather, in reality they cool down the summer temperatures in the CSSC. It was stated previously that effluents from the MWRDGC plants constitute most of the flow during low flow periods and the stream is effluent dominated. Thus the temperature of the effluents constitutes the base temperature of the river, more so than a natural temperature of the river. The stream receives some natural flow from the upstream Des Plaines River.

Thermal Standards

The Clean Water Act defines thermal loads that exceed standards commensurate with Section 101 of the Clean water Act as thermal pollution and thermal standards have been implemented in every state. Thermal/temperature standards of the state of Illinois were summarized in Table 2.1 and restated below.

General Use

The General Use numeric standards require the water temperature to be less than or equal to 32°C (90°F) for the months of April to November and 16°C (60°F) for the remaining months of the year. These limits cannot be exceeded for more than 1% of the hours in the 12-month period ending with anymonth. The maximum deviation during this allowed exceedance time is 1.8°C (3°F), meaning that the maximum temperature that cannot be exceeded is 93°F (34 °C). The narrative standards are:

- There shall be no abnormal temperature change that may adversely affect aquatic life unless caused by natural conditions.
- The normal daily and seasonal temperature fluctuations which existed before the addition of heat due to other than natural causes shall be maintained.
- The maximum temperature rise above the natural temperature shall not exceed 2.8°C (5°F).

The 1% allowable excursion time limit represents approximately 88 hours.

The General Use Standards are in force at the end of the investigated reach at the I-55 bridge and further downstream. The Illinois Pollution Control Board granted the Commonwealth Edison (the predecessor of the Midwest Generation) an Adjusted Standard that is applicable to the location at the I-55 bridge. (Based on the communication from Midwest Generation, Commonwealth Edison once held a thermal variance which covered the entire waterway from the I-55 Bridge down to the confluence of the Des Plaines River with the Kankakee River. This variance was commonly known as the "Five Mile Stretch Variance." However, it has not been in effect since the mid to late 1980's.). The Adjusted Standard for the I-55 bridge is as follows:

January	16°C (60°F)	February	16°C (60°F)
March	18°C (65°F)	April 1-15	23°C (73°F)
April 16-30	27°C (80°F)	May 1-15	29.5°C (85°F)
May 16-31	32°C (90°F)	June 1-15	32°C (90°F)
June 16-30	33°C (91°F)	July	33°C (91°F)
August	33°C (91°F)	September	32°C (90°F)
October	29.5°C (85°F)	November	24°C (75°F)
December	18°C (65°F)		

These standards may be exceeded by no more than 3°F (1.7°C) during 2% of the hours in the 12-month period ending December 31, except that at no time shall Midwest Generation's plants cause the water temperature at the I-55 bridge to exceed 93°F (34°C). The 2% allowable excursion time limit represents approximately 175 hours

Secondary Contact and Indigenous Aquatic Life Use

Temperature shall not exceed 34°C (93°F) more than 5% of the time, or 37.8 °C (100°F) at any time. The 5% allowable excursion time limit represents approximately 438 hours.

History of the Standard

The Illinois Pollution Control Board at its deliberation of March 7, 1972 defined the General Use for the state's waters. The Board at this meeting adopted the dissolved oxygen standard of 6 mg/L for16 hours and the absolute minimum of 5 mg/L and repealed the previous standard (5 mg/L and 4.0 mg/L).

The Board also defined so called "restricted waters" that later were changed to the "Secondary Contact and Indigenous Aquatic Life Use" to include federal wording defined for these waters. These waters were to include *"certain additional heavily industrial channels in the Chicago area. The evidence establishes that even with the most advanced treatment and with stormwater overflow control aquatic life standards for dissolved oxygen (and perhaps also ammonia) cannot be met in portions of the Chicago River Systems, and that <u>meeting the aquatic temperature standards in these same areas, as well as in the adjacent sections of the Des Plaines River, would require cooling towers costing tens of millions of dollars and produce doubtful benefits in terms of stream improvement</u>"*[8]. The Board decided that the I-55 bridge is the dividing point between on the Des Plaines River between the upstream "restricted" (secondary contact and indigenous aquatic life use) and downstream general use.

On November 8, 1973 the Illinois Pollution Control Board conducted hearings on the establishment of the Illinois water quality standards (Appendix A). Temperature standards received considerable attention. At the hearing, Commonwealth Edison proposed an amendment to loosen the temperature standard below the I-55 to its confluence with the Kankakee river. In response to the request of the hearing officer to tighten up its proposal to reflect the minimum temperatures possible, Commonwealth Edison withdrew its original proposal and substituted an amendment to Section 203(i)(4) in Chapter 3 of the Pollution Control Board's Water Pollution Regulations which proposed individual monthly temperature limits for the I-55 section of the Des Plaines River and, consequently, for the 5-mile stretch between I-55 and the confluence with the Kankakee River. The Board adopted the final Edison amendment as published with exceptions. It set 90°F (32°C) as the maximum temperature standard for the months of July and August and reduced the excursions to 4% of the previous twelve-month period. The excursion would allow up to 14.6 days per year for the

[8]Underlined by the authors of this UAA.

temperature to be higher by as much as 5°F (3°C). <u>*The Board also set an automatic termination date of July 1, 1978 at which time the General Use standards would have applied again*</u>. Appendix A contains the record of the hearing and summary of the discussion and presentation of the Commonwealth Edison as well as opposing views by the USEPA, Illinois environmental agencies and private citizens. *The above variance expired a long time ago and has not been used for more than ten years; therefore, the discussion in this report is included only for historical purposes.*

The following discussion is based on the memorandum by Connie Tonsor of the Illinois EPA describing the more recent development (included in Appendix A):

> *On June 19, 1987. Commonwealth Edison filed an amended petition for a thermal demonstration. In September 1987, the Illinois Pollution Board asked Commonwealth Edison to prove that their discharges do not adversely affect the general use waters (i.e., downstream of I-55 bridge. On November 15, 1989, the Board found that Commonwealth Edison successfully made the demonstration. The Board noted that Commonwealth Edison and the Illinois EPA agreed that heat was not a factor limiting the quality of the aquatic habitat of the Five-Mile stretch. During the proceedings, the Illinois EPA supported Edison's conclusion that the discharge complied with both the secondary contact and General Use standards. The Board noted that the Agency (IEPA) concluded that as long as the Joliet Station meets all the applicable standards at the point of discharge and in the downstream General Use waters, the Agency did not view the Joliet Station's thermal discharges as limiting aquatic diversity in the receiving waters.*

> *On November 21, 1991, the Board granted Commonwealth Edison a variance from the requirements of 35 Ill. Adm. Code 302.211(d) and (e) to conduct a study of the Upper Illinois Water Way and the impact of heated effluent discharges to the receiving stream. The study then would become the basis of an adjusted standards/alternate thermal standard, if needed. Edison subsequently conducted an extensive and exhaustive study on the thermal effects caused by the heated discharges (Commonwealth Edison, 1996). This study was conducted by a reputable team of scientists from three universities (DePauw University, Iowa Institute of Hydraulic Research and Wright University) and Edison ecological consultants.*

> *On May, 1996, Commonwealth Edison filed a petition for adjusted thermal standards for I-55, as listed above. On October 3, 1996 the Illinois Pollution Control Board granted the adjusted standards for the I-55 location as specified in the preceding section. These alternate standards were granted on a premise that " the cost of additional cooling may not be economically reasonable when compared to the likelihood of no improvement in the aquatic community of the UIW " (AS96-10, information in Midwest Generation, 2003).*

On March 16, 2000, the Pollution Control Board granted the transfer of the Adjusted I-55 Thermal limitations from the Commonwealth Edison to Midwest Generation in AS 96-10, with concurrence of the Illinois Environmental Protection Agency and with no opposing views by US EPA or private parties presented.

Mixing Zone Issues

Rule 302.102 of the Illinois Administrative Code, Title 35 defines Allowed Mixing and Mixing Zones. Such mixing zones are allowed provided that the discharger complies with the requirements of 35 Ill. Ad Code 304.102. This rule states *that dilution of the effluent from the treatment works is not acceptable as a method of treatment of wastes in order to meet the standards* and requires the discharger *to provide the best treatment of wastewater consistent with the technology feasibility, economic reasonableness and sound engineering judgement.*

Several issues of the Rule 302.102 should also be noted that are pertinent to the Lower Des Plaines River (numbers reflect numbering of paragraphs in the Rule):

7. The area and volume in which mixing occurs, alone or in combination with other areas and volumes of mixing, must not intersect any area of any body of water in such a manner that the maintenance of aquatic life in the body of water as a whole would be adversely affected.
8. The area and volume in which mixing occurs, alone or in combination with other areas and volumes of mixing must not contain more than 25% of the cross-sectional area or volume of flow of a stream except for those streams where the dilution ratio is less than 3:1.
10. No body of water may be used totally for mixing of single outfall or combination of outfalls.
11. Single sources of effluents which have more than one outfall shall be limited to a total area and volume of mixing no larger than that allowable if a single outfall were used.

The reasoning and limitation of the mixing zone for thermal effluents have been discussed and established more than thirty years ago (for example, see presentation by the Director of Water Quality Standards Section of the FWPCA S. Burd (1969)). Burd specified that the passage, i.e., the zone not affected by excessive temperature, should be at least 75 percent of flow or cross-sectional area. Apparently this requirement was incorporated into the Illinois Administrative Code. The regulation is unclear how the mixing zone can be applied to situations when the dilution flow is below 3:1 ratio as it is common with the thermal discharges in the Lower Des Plaines River.

We were informed by the Illinois EPA and Midwest Generation that the mixing zone is applicable and was included in the discharge permit. The maximum size of the surface area of the mixing zone was set as 26 acres; however, as the rule mandates, only 25% of the cross-sectional area or flow is available for mixing when the dilution ratio is greater than 3:1. Acute toxicity is allowed only within a zone of initial mixing; chronic toxicity is allowed within the mixing zone. In general, we find the mixing rule sound, provided that the 75% passage zone is implemented.

We are aware of the studies conducted by the Midwest Generation to clarify establishing the mixing zone and negotiation with authorities regarding the extent of the mixing zone especially at flows that do not provide greater than 3:1 flow to discharge ratio. This issue is outside of this UAA.

Water Body Assessment for Temperature

The Commonwealth Edison (1996) (currently Midwest Generation) reports gathered excellent information on the temperature, physical and biological data on the Upper Illinois Waterway. This UAA is using, as much as possible, the information from this research effort. We are aware that there was to be a concurrent submission of a proposal for alternate UAA or a variance from Midwest Generation. However, as a part of Water Body Assessment, we will address the issues of temperature and temperature standards. Our reference level is the statewide General Use standard for temperature or the Alternate General Use standard and not the Secondary Use and Indigenous Aquatic Life standard. In other words we will be asking in this UAA whether the General Use standard for temperature is attainable and then we will test appropriateness of the Secondary Use standard. If the General Use standard is attainable, any potential deviations (including the Secondary Use standard) should be tested using the 6 Reasons outlined in Box 1.1.

Compliance of Temperature with the Standing General Standards

Temperature has been measured at several sampling locations either as a part of the sample collection effort on the day of sampling or continuously at the I-55 bridge by the Midwest Generation. No continuous temperature measurement is performed in the stretch of the Upper Dresden Island pool between the Joliet thermal discharges from the power plants and the I-55 bridge that would enable one to directly assess compliance with the Secondary Contact-Indigenous Aquatic Life standards in this stretch. Midwest Generation; however, continuously measures temperatures at the discharge outlets of the cooling water from the two Joliet power producing units. The capacity flow requirements of the power plant units (reported in Table 1.2 in the preceding chapter) exceed the design (near the 7-Q-10) low flow in the river. Therefore, during the river flows that are near or less than the condenser flows, most--if not all--flow in the river could be taken by the power plants, unless Midwest Generation uses production cutbacks and reduces the demand on cooling water during the low flow. Based on the information provided by Midwest Generation, production cutbacks and condenser flow reduction do occur under these circumstances.

Midwest Generation also provided the following information on cooling in the discharge canal of the Unit # 29: Joliet Station #29 uses 24 mechanical draft cooling towers to dissipate the heat in the discharge canal prior to its entry into the Lower Des Plaines River. The towers are designed to cool from 1/3 to ½ of the total condenser flow of the Joliet Station #29. The design ΔT on the towers is 14°F, and monitoring by Midwest Generation over the past several summers shows much higher values and, therefore, greater efficiencies in dissipating heat. When all 24 cooling towers are operating, the condenser discharge temperatures are cooled by an additional 5°F or more before combining with the main body of the river.

Figures 2.37 to 2.40 show probability distributions of temperatures at the MWRDGC and IEPA monitoring stations on the Lower Des Plaines River. Temperature is not a priority pollutant; therefore, the 99.8 percentile decision point for comparison with the General Use Standard does not apply. As a matter of fact, none of the grab measurements have exceeded the standard of 32°C. Figure 2.41 is probability of temperature at the reference location on the Kankakee River at

Momence. Statistical probability plotting for temperature is a way to present data. This type of data presentation and plotting has been routinely used in the past assessment of the Des Plaines Rivers (e.g., MWRDGC reports by Butts et al., 1975).

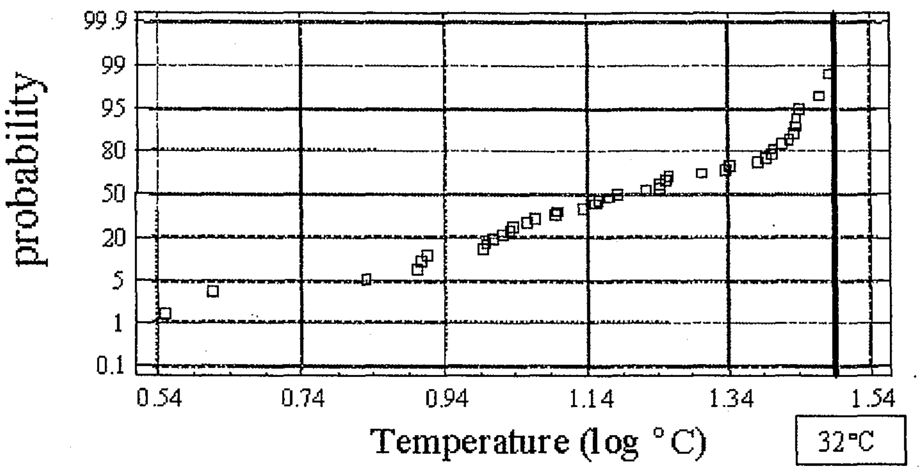


Figure 2.37 Statistical Probability Plot of Temperature at IEPA G-23 in Joliet - Brandon Pool

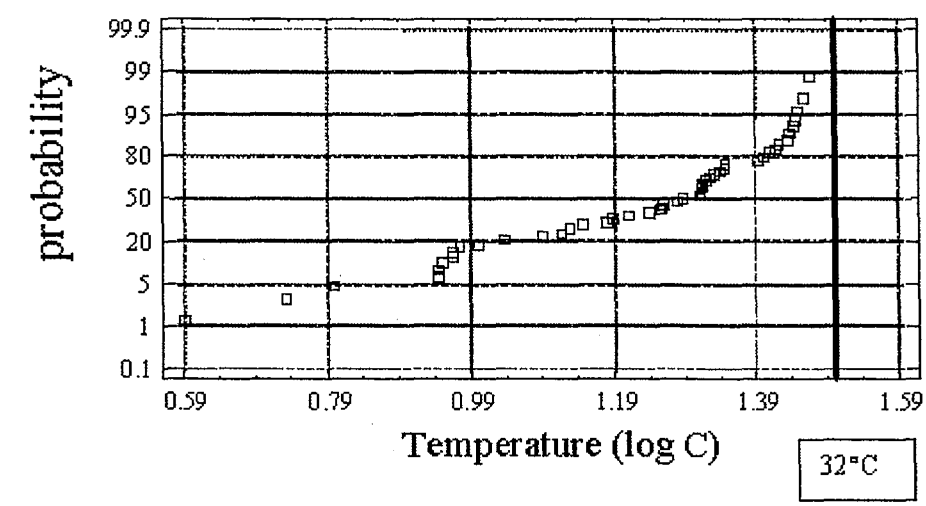


Figure 2.38 Statistical Probability Plot of Temperature at MWRDGC 93 in Joliet on Brandon Pool (years 2000-2001)

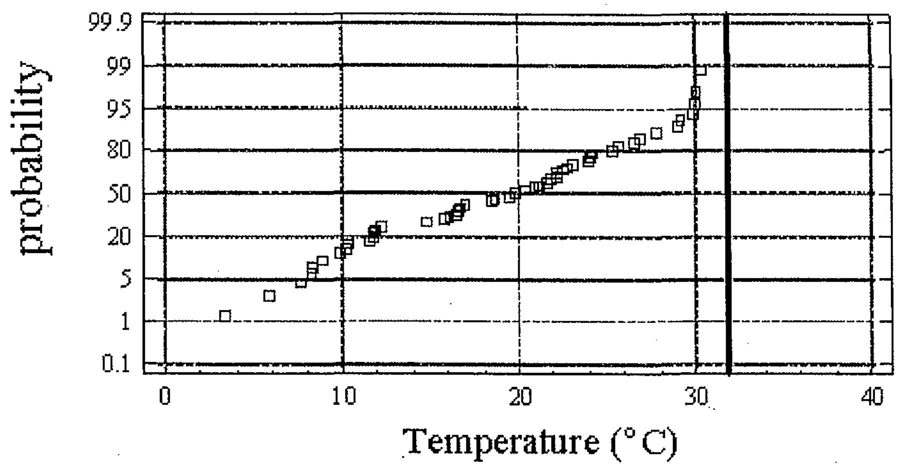


Figure 2.39 Statistical Probability of Temperature at MWRDGC 94 - Dresden Island Pool - Empress Casino (years 2000 - 2001)

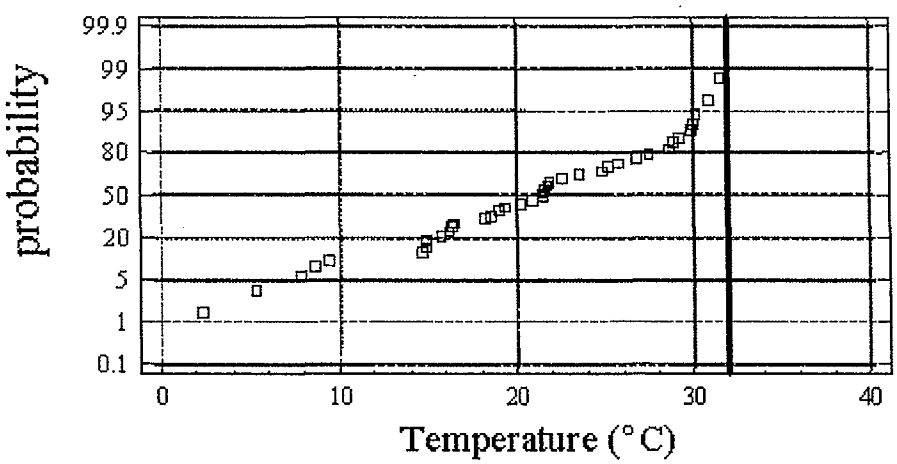


Figure 2.40 Probability Distribution of Temperature at MWRDGC95 - I-55 Bridge - Dresden Island Pool

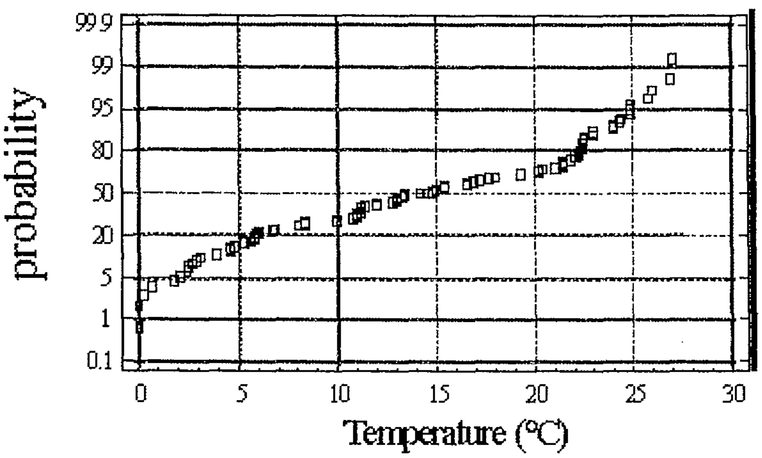


Figure 2.41 Probability Distribution of Temperature for Kankakee River at Momence - Reference Stream

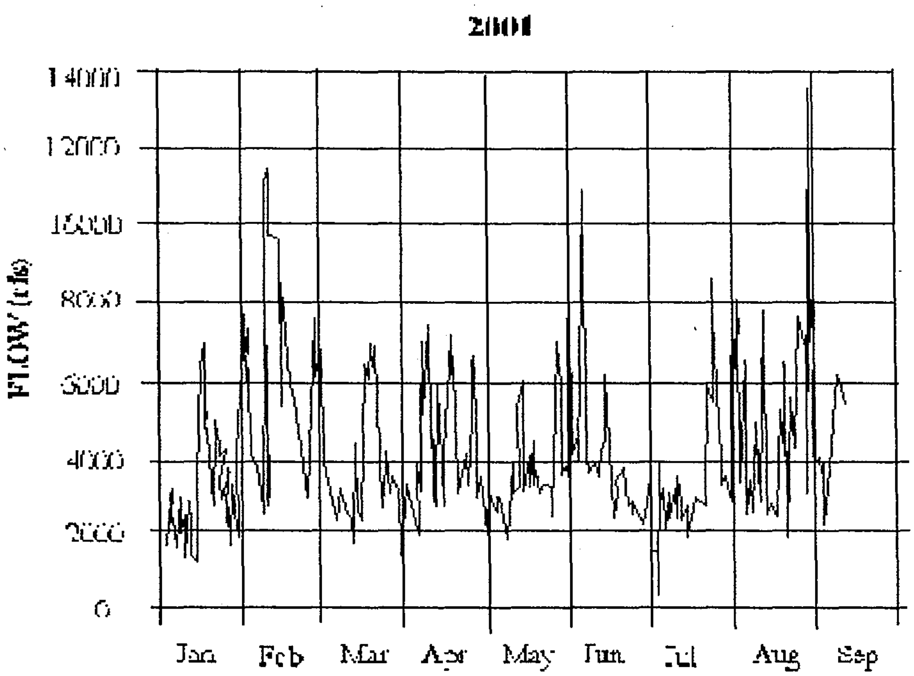


Figure 2.42 Flows in the Des Plaines River at the Brandon Road Lock and Dam upstream of the Joliet Power plants, in 2001. Measured by the U.S. Army Corps of Engineers.

Figure 2.42 shows river flows upstream of the Joliet plants for the year 2001. The figure documents that in late June-beginning of July 2001 period, flows were at the level approaching or even less than the magnitude of the capacity condenser flow from the two Joliet units.

The reference water temperatures on Figure 2.41 are well below the 32°C (90°F) standard. However, it should be pointed out that the MWRDGC Station 94 and 95, located in the Dresden Island Pool contain data for the years 2000- 2001 only. As it will be subsequently shown, measured temperatures during 1999 at the I-55 bridge and in the discharge channels by the Midwest Generation were higher than in the 2000-2001.

Type of Cooling at the Joliet Plants

The type of condenser cooling installed at the Joliet power plants is once-through cooling. In this type of cooling, water is withdrawn from the river, passes the condenser in the cooling system, and is then--with added heat--returned back to the river without recycling. The added heat results in an increase of water temperature in the receiving water body and the heat is then dissipated by the receiving water body or carried downstream. If the flow of the river is about the same as the cooling water flow, as it would be in the case during low flow on the Lower Des Plaines River, the temperate increase before and after the power plant is about the same as the temperature difference in the cooling water intake and discharge channels. Information provided by the Midwest Generation and presented in Table 1.2 specified the ΔT through the condensers as being 9.4°F (5.2 °C) at design flow.

The temperature difference in the river before and after the thermal discharge obviously depends on the magnitude of flow. If the flow was at the 7Q10 level (1950 cfs), it would be significantly less than the cooling water requirement of the plants reported as 2620 cfs. Then a part of the heated discharge may be forced by created back currents back into the intake, thus increasing the temperature downstream from the plant even further. Flow in the river greatly fluctuates due to the operation of the CSSC and upstream Lockport and Brandon Road Dam locks (Figure 2.42).

An alternative to the once through cooling used at the Joliet plants is a closed recycle cooling with natural draft or mechanical cooling towers (for example, the WE power plants near Portage and Kenosha, Wisconsin) or lakes (Dresden plant) that result in less discharge flow, typically 2 - 4% of a comparable once through cooling system, with a commensurate smaller heat load on the receiving water body. As stated previously the utility has installed (prior to purchase by the Midwest Generation) 24 mechanical draft cooling towers capable of cooling approximately one-third of Joliet #29 total discharge flow. These towers are located on the discharge channel of the Unit #29 and do not allow recycle. The cooling towers are used on an as-needed basis.

As stated in the Midwest Generation presentation to the biological subcommittee, the use of the existing cooling towers alone is often not sufficient to control the magnitude of the thermal discharge to meet current near and far-field limits and the utility has to use plant production derating (i.e., forced production cutbacks) to meet the existing standard.

Selection of the Temperature Standard

Excessive temperature is pollution (added excess heat is a pollutant), stimulant, catalyst, depressant, shortly, one of the most important and most influential water quality *characteristics*. When the 1972 amendments of the Clean Water Act were formulated, *thermal pollution* has received considerable attention from the scientific community and environmental officials, and received special attention in the CWA.

The USEPA (1986) Criteria document contains extensive discussion on effects and impacts of the increased temperature and thermal pollution. The following common knowledge effects of temperature on the integrity of the receiving water body are known or reported in literature and have been also summarized in the USEPA (1986) water quality criteria document and in Krenkel and Novotny (1980):

1. High temperature has acute and chronic toxicity effects on aquatic organisms (negative common knowledge effect, see US EPA (1986)).
2. Temperature increases chemical and biochemical reaction rates in the water body such as decay rate of biodegradable organic matter, sediment oxygen demand (SOD), nitrification, reaeration (supply rate of oxygen from the atmosphere into water) (both negative and positive common knowledge effects, see USEPA, 1986)).

   Positive: increasing decomposition of organic dissolved and particulate matter in water and sediments

   Negative: (a) Increased SOD

   (b) optimum temperature for nitrification (converting ammonium to nitrate in water and top layer of sediment) is 22°C and rate of nitrification decreases significantly with further increase of temperature (Zanoni, 1969). This may result in an increased ammonium release from diagenesis (anaerobic breakdown of organic particulate carbon) in sediments (DiToro et al., 1990; DiToro, 2000) whereby ammonium released from the sediment is nitrified in the upper aerobic sediment layer.
3. Temperature decreases dissolved oxygen saturation values and DO solubility (see the preceding section on dissolved oxygen), consequently less oxygen can be dissolved in the river from the atmosphere (negative effect) and, in some instances involving high temperature, oxygen can be lost.
4. Temperature affects the biological processes such as growth and nourishment of the aquatic organisms, decomposition of organic matter in water and sediments, photosynthesis and respiration of algae and macrophyte aquatic plants, and dye-off of pathogenic micro-organisms, viruses and indicators of pathogenic pollution (fecal coliforms) (both positive and negative common knowledge effects, see, for example Thomann and Mueller, 1987).
5. Temperature increases chronic toxicity of ammonium and other toxic compounds (USEPA, 1986; 1999). Consequently, the magnitude of the chronic CCC standard for ammonium is decreasing with temperature. The CCC standard is related to temperature (see USEPA, 1999).

6. It affects the comfort of swimmers (the comfortable range of temperature for swimming ranges from 25°C to 30°C) (common knowledge, see also USEPA, 1986).
7. It impacts the fish in the following ways (Krenkel and Novotny, 1980, USEPA, 1986)
    1. Direct death from excessive temperature rise beyond the thermal lethal point
    2. Indirect death due to
        1. Less oxygen available
        2. Disruption of food supply
        3. Decreased resistance to toxic materials
        4. Predation from more tolerant species
        5. Synergisms with toxic substances
        6. Decreased resistance to disease
    3. Decrease in respiration and growth
    4. Competitive replacement by more temperature tolerant species
    5. Sublethal effects
8. The number and distribution of bottom organisms decrease as temperature increase. The upper limit for a balanced benthic population structure is approximately 32°C (90°F)(USEPA, 1986).
9. It changes the algal composition, shifting algae in higher temperatures to more problematic blue-green species (see Figure 2.43 replotted from Cairns, 1955). For example, from 20°C to 25 °C, diatoms predominated, green algae predominated from 30 °C to 35 °C, and blue - greens predominate above 35 °C. Algal blooms made of blue - green algae release toxins that are harmful to swimmers and prevent contact recreation (Carmichael et al., 1985).

The above statements and concerns are pertinent for a formulation of a long term thermal standard and may not reflect the current situation of the Des Plaines River. For example, ammonium is not currently a serious problem (with exception of potential sediment toxicity - see Chapter 3) and, if it became one in the future, the focus would be on identifying and remedy of the increased ammonium loads. Also excessive occurrence of blue green algae has not been observed based on the biologic studies performed by the Midwest Generation. The effects of increased temperature on the biotic integrity of receiving waters can depend on numerous factors, such as season of the year, trophic status of the ecosystem, levels of toxins, among others.

Earlier in the water quality standards development, standards were defined in terms of avoiding lethal levels. In current water quality standards guidelines and regulations, water quality standards are developed, formulated and implemented to protect the potentially indigenous biota in the water body. The term potential is important. If the water body is stressed and the biota has been adversely affected by pollution (thermal or chemical) or other effects (pollution in this context is understood according to the definition of pollution in Section 5 of the CWA), the standards should not be developed to protect the inferior biotic composition. The standards should also contain some margin of safety that the US EPA criteria guideline document specifies as 2°C (3.6 °F).

For example, the USEPA guidelines and water quality standards regulations require that standards are developed to protect 95% of indigenous organisms with a margin of safety set at about one half of the Final Acute Value. This approach may not be directly applicable to temperature. No one

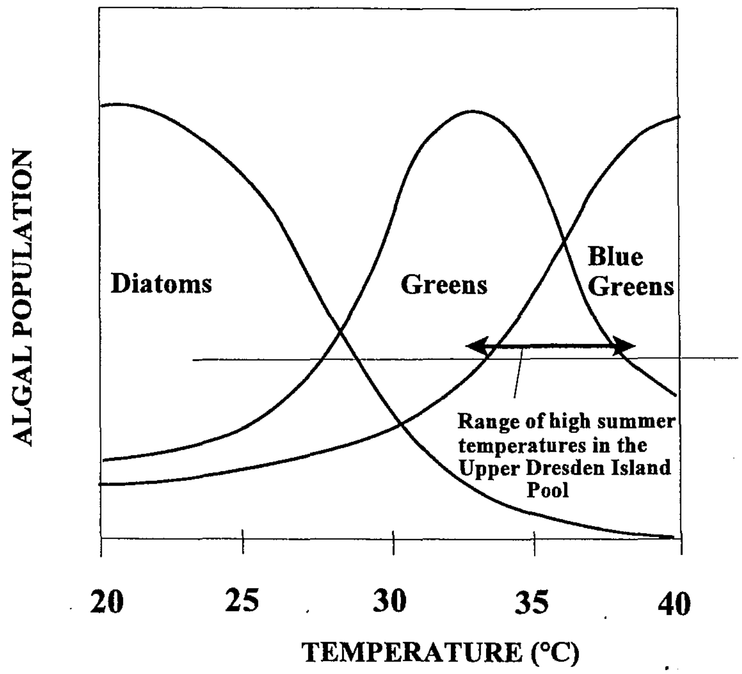


Figure 2.43 Algae Population Shift with Change in Temperature (Cairns, 1955). Lower part of the range is typical for the reach upstream of I-55, higher temperatres are measured near the discharge canals.

would assign a temperature standard at ½ of the lethal value because the preferred optimal temperatures may not be far below the lethal temperature. However, no standards should be proposed and accepted that would be above a lethal limit.

The USEPA (1986) water quality standards define two upper limiting temperatures for a location:

1. One limit consists of a maximum temperature for short exposures that is time dependent and is given by a species-specific equation (see USEPA, 1986).
2. The second value is a limit on weekly average temperature values that:
   a. In the cooler months (mid-October to mid-April in the north) 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°C;
   or
   b. In the warmer months (April through October) is determined by adding the physiological optimum temperature (usually for growth) a factor calculated as one-third of the difference between the ultimate upper incipient temperature for the most sensitive important species that normally is found at that location and time.
   or
   c. During reproductive seasons (generally April through June and September) 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.

or

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

The current Illinois General Use thermal standards comply with the USEPA (1986) standards recommendations.

Critique of the Current Secondary Contact and Indigenous Aquatic Life Standard

From the records of the hearings in 1972 and 1973, presented in the preceding section and in Appendix A, it is apparent that Illinois Secondary Contact and Indigenous Aquatic Life standards were implemented and accepted by the Illinois Pollution Control Board based on the benefit-cost analysis and to avoid cost of cooling on the Lower Des Plaines River that was perceived as hopelessly polluted. In the subsequent years, water quality of the river has improved dramatically, both chemically and biologically. After evaluating all data, it is our belief that the river can continue to improve and reach its ecological optimum that would be commensurate with the goals of the Clean Water Act. Standards that are not in compliance with Section 101(a) of the CWA must be addressed by the UAA.

The first question to be addressed is whether the current General Use or Secondary Contact and Indigenous Aquatic Life standards are protective of the indigenous aquatic biota that is or could be residing in the Lower Des Plaines River. The USEPA (1986) temperature criteria guidelines presented formulae for calculation of the above thermal limits for development of statewide or water body specific standards. They also specify that to provide a safety factor so that none or only a few organisms will perish, a standard should be set 2°C below maximum temperature.

Eaton et al. (1995) published the upper thermal tolerance limits for fish as follows:

Warmwater species	Upper lethal limit °C (°F)	Max 95% Tolerance Limit
Gizzard shad	36.5 (97.7)	31.5(88.7)
Common Carp	36 (96.8)	31.4(88.5)
Channel Catfish	37.8 (100)	31.6(88.8)
Largemouth Bass	36.4 (97.5)	31.7(89.1)
Bluegill	37.3 (99.1)	29.5(85.1)
Smallmouth Bass	35 (95)	29.5(85.1)
Freshwater Drum	32.8 (91)	32.4(90.3)
Golden Shiner	34.7(94.4)	30.8(87.4)
Green Sunfish	35.4(95.7)	31.7(89.0)

The first column in the above table represents maximum tolerable limit of a short duration exposure (1 to 7 days) after acclimation measured in the laboratory by various authors (referenced in Eaton et al. (1995)). This implies that if temperatures exceed this limit fish will not survive even with acclimation and in laboratory conditions where other stresses are not present. The second column represents data based estimation of 95% tolerance of fish of a given species to maximum average weekly temperatures. Obviously, the 95% limit based on average weekly temperatures is less than the absolute laboratory short exposure maximum after acclimation; however, it provides a better information on actual natural thermal tolerance and reflects the rationale of developing standards that would provide 95% protection of most sensitive indigenous species. Figures 2.44 and 2.45 present the plot of the range of lethal temperatures found in literature. Data for Figure 2.44 were provided by the Midwest Generation in their presentation to the biological subcommittee for this study. Figure 2.45 contains data compiled by the US Fish and Wildlife Service. We have added lines representing the absolute limits of the chronic zone of the standard. It should be noted that the General Use standard allows the temperature to be in the chronic to low acute zone for about 3½ days, the Adjusted standard for I-55 allows about 7 days, and the Secondary Use and Indigenous Aquatic Life Standards allow temperature to be in the chronic to acutely lethal zone for 18 days.

The selected species on these figures are representative of the warm water fish species that have been found and/or could potentially live in the Lower Des Plaines River. We have plotted the summer General Use, Alternate I-55 General Use and Secondary Contact and Indigenous Aquatic Life Illinois maximum standards on these charts.

We found that the Secondary Contact Indigenous Aquatic Life maximum standard exceeds the lethal limit for most indigenous adult fish species. **Without even considering the required margin of safety of 2 C required by the USEPA(1986) criteria document, the maximum lethal standard should have been set at about the level commensurate with the current statewide General Use standard, i.e., 34 C (93 F) and current alternate maximum standard limit. The Secondary Use Indigenous Aquatic Life acute standard of 37.8 C (100 F) is lethal and provides no protection.**

We will summarize the contradiction of the Secondary Use and Indigenous Aquatic Life Standards in Chapter 8 where we will point out the differences between the Objective of the Standards

-supporting an indigenous aquatic life limited only by the physical configuration of the body of water-

and the lethal magnitude of some standards for this use listed in Table 2.1.

Experiments by Wright University to Establish Temperature Limits. The team from the Wright University headed by Dr. Burton was retained by the Commonwealth Edison to study the temperature effects in the Des Plaines River on the biota. Earlier results of the studies were presented in a report (Burton, 1995) and more recent results in memoranda (Burton et al., 1998; and Burton and Rowland, 1999). The significance of the report is that the work and studies were performed on the site and/or with water and sediments taken from the Des Plaines River. The 1995 report focused on the sediments and will be discussed in more detail in Chapter 3. Of note to this chapter are the result of a bioassay in which a fish *(fathead minnow -Pimephales promelas)* and benthic invertebrate Scud (*Hialella azteca)* were exposed to site water and water with a contaminated sediment taken from the river. The experiment showed that survival of Scud was relatively high (80 - 90 % in water and 40 to 75% in water with sediment) at temperatures 25 to 30 °C but only 20 % of organisms survived at a temperature of 35 °C. For fish the survival was 50 to 80 % at temperatures 20 to 30 °C but 0% at

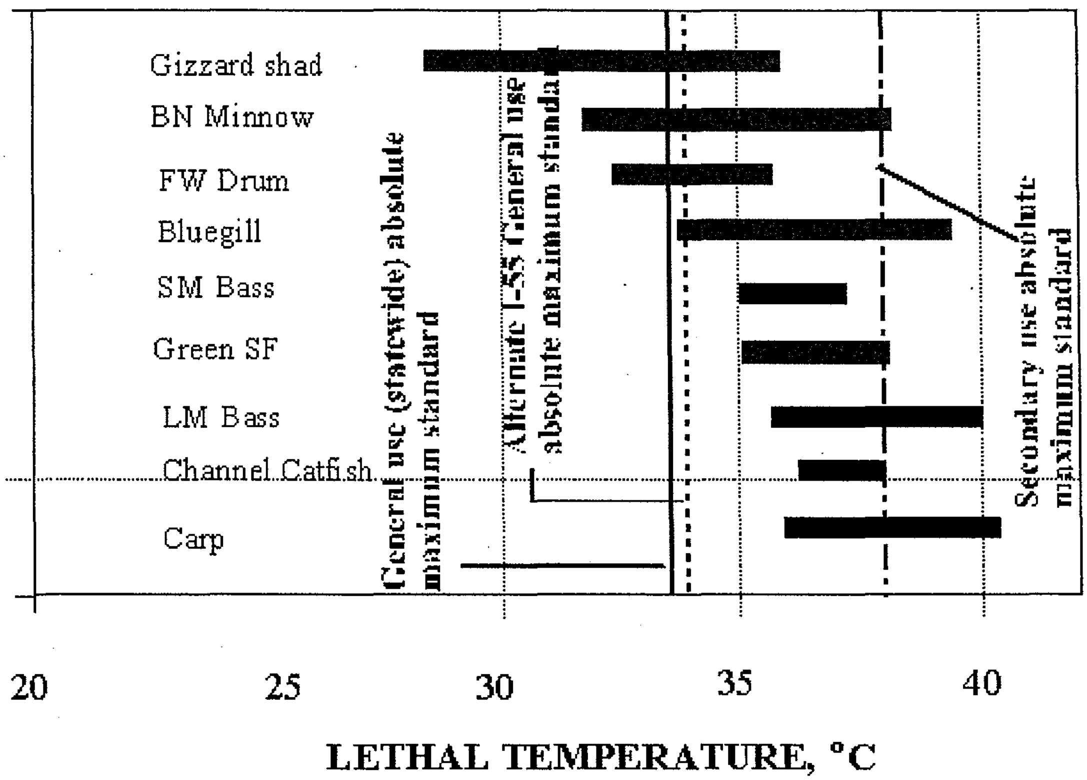


Figure 2.44 **Comparison of lethal temperatures and the current temperature standards for the Lower Des Plaines River. Data on lethal temperatures provided by Midwest Generation to the biological subcommittee of the Lower Des Plaines River Use Attainability Analysis and included also in the Summary Report (Midwest Generations, 2003)**

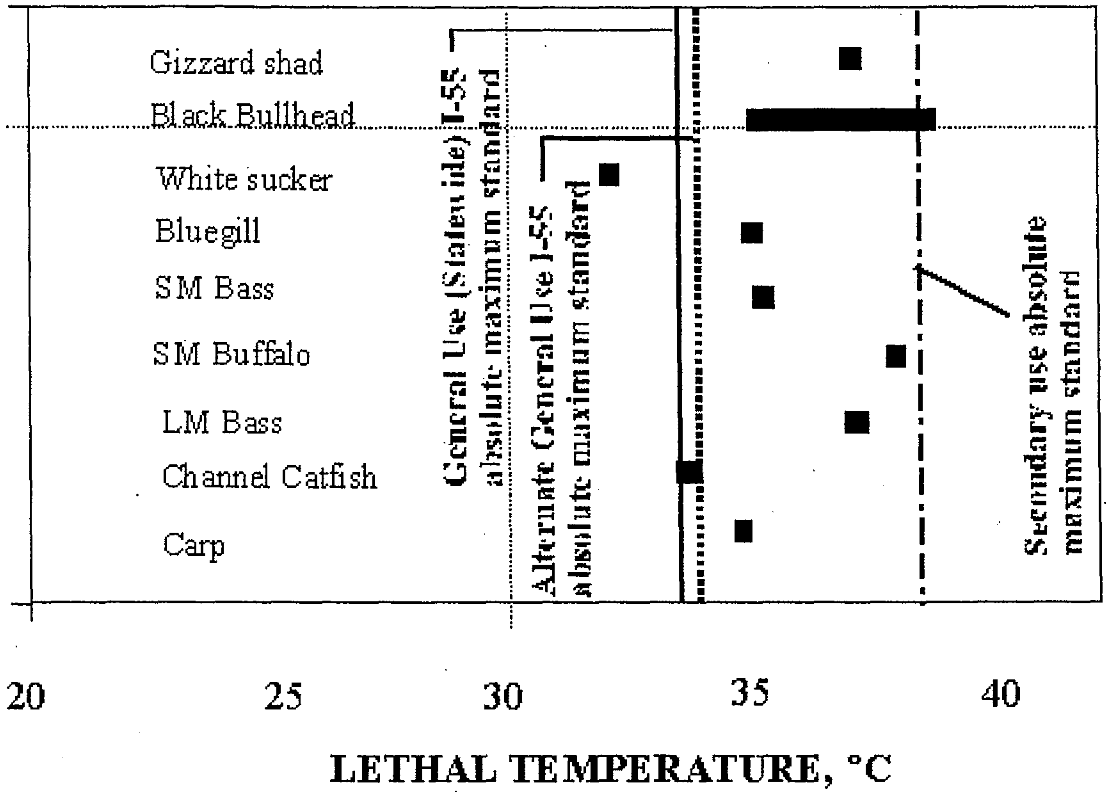


Figure 2.45 Lethal temperatures and standards. Data from U.S. Fish and Wildlife Service database taken from Banner and Arman, 1973; Block, 1952; Brungs and Jones, 1977; Cvancara et al., 1977; Brett, 1944; Carlander, 1969; Cherry et al., 1975; Horning and Perason, 1973; Larimore and Duever, 1968; Leidy and Jenkins, 1977; Cambell and Branson, 1978; Miller, 1960; Meuvis and Heuts, 1957.

temperature of 35°C. This bioassay indicated that the lethal threshold is between 30 and 35°C (86 to 95 °F). The sediment used in the experiments was taken from the accumulation of the highly contaminated sediment just upstream of the Brandon dam that does not represent the sediment contamination level in the Dresden Pool affected by the thermal discharges (see Chapter 3).

In the more recent study described in the 1998 and 1999 memoranda (Burton et al., 1998; Burton and Rowland, 1999), continuous in-situ biomonitoring was performed (1) in the discharge canal of the Unit # 29 during summer conditions, (2) simultaneously in the artificial stream using similar *in situ* assays, (3) intensive 7 day ammonia and temperature study to measure ammonia variation in sediments and overlying water; and (4) thermal effects characterization of 3 species at temperatures ranging to 93 °F (34.4 °C) over a period of 7 days in controlled laboratory experiments. The test organisms included fish fathead minnow (*Pimephales promelas);* aphipod Scud (*Hyalella azteca),* possible. Such thermal differential standard is applied to the upstream and downstream temperatures. The notion of natural temperature is typically included for cases when the natural temperature itself may get higher.

Conclusion on Temperature

Temperature is one of the more significant parameters being addressed in this study, particularly within the Dresden Island pool. Temperature has been repeatedly addressed by the Pollution Control Board since the original standards were established in 1973 and as recently as 1996. In light of significant operational and financial impact thermal standards have on Midwest Generation's facilities; Illinois EPA requested that this analysis addresses two specific issues and defer a recommendation on proposed future standards such that Midwest Generation and other river users could contribute to the socio-economic factors. A socio-economic analysis and determination whether the impact on the dischargers of heated effluents on the Lower Des Plaines River would incur a substantial and wide spread adverse socio-economic impact on the utilities and the population was not performed in this study but is crucial. It is the only reason a departure from the Illinois General Use standard can be justified. This study has concluded that the first five reasons by themselves, cannot be applied to downgrade the thermal standard from that specified by the Illinois General Use standards.

The two specific issues addressed to be addressed in this UAA are:

1) determination of whether current thermal conditions are detrimentally impacting the aquatic community that inhabits the study reach; and

2) determination of whether currently applicable state standard (Secondary Contact and Indigenous Aquatic Life standards modified for the Dresden Pool) is adequate to protect the aquatic community otherwise capable of inhabiting the study reach.

If a negative conclusion results in either instance and if it is found that the implementation of the General Use Standard would cause a substantial and wide spread socio-economic impact, it is recommended that the Agency collaborates with the stakeholders group, particularly Midwest Generation, to devise and propose new thermal standard that would be both environmentally protective and financially and technically attainable.

Through the review presented in this chapter and the underlying data, we concluded the following:

- Ammonium chronic toxicity in water and sediments is increased as a result of temperature. High temperature affects the ammonium toxicity directly by making it more toxic and, by reducing nitrification in the upper sediment layer, it causes more release of ammonium from the sediment.

- The high temperature could cause a shift of the algal population in this nutrient enriched stream to undesirable blue green algae that produce undesirable toxins harmful to swimmers.

- High temperature reduces reaeration capability of the stream by reducing the oxygen saturation to values of about 6 mg/L at temperatures at 37.8°C (100°F)
- The currently applicable maximum thermal standards are higher than lethal ranges in published literature for species indigenous to the area and demonstrated to be tolerant to other environmental conditions existing within the upper Dresden Island pool.

- Current temperature standards for the Lower des Plaines River are also higher than allowable temperatures in virtually all other states.

- Current temperature standards allow longer periods of high temperature (up to 18 days) to be in the acutely lethal zone.

Because the existing thermal standards for the Lower des Plaines River allow the temperatures to reach lethal levels and stay there for an extended period of time we have concluded that the current Secondary Use and Indigenous Aquatic Life temperature standards do not provide adequate protection to the indigenous and potentially indigenous aquatic organisms and should be replaced by a standard that equals or is close to the statewide General Use temperature standard.

Brief Evaluation of the Six UAA Reasons for Temperature

(1) Naturally occurring pollutant concentrations prevent attainment of the use.
Elevated temperatures in the Dresden Island Pool are not natural. This reason does not apply.

(2) Natural, ephemeral, intermittent or low flow or water levels prevent the attainment of the use unless these conditions may be compensated for by the discharge of a sufficient volume of effluent discharge without violating state conservation requirements to enable uses to be met.
This reason does not apply. The flow in the river is increased by diversions.

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place.
Reducing temperature would improve biotic integrity of the Lower des Plaines River.

(4) Dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use.
The reason does not apply. Impounded water bodies are not exempt from General Use unless the conditions cause an irreversible physical impairment of the habitat (e.g., Brandon Pool). Such conditions do not exist in the Dresden Island Pool.

(5) Physical conditions related to the natural features of the water body, such as the lack of proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses.

The reason does not apply. The Upper Dresden Island Pool is physically similar to other pools on the Upper Illinois River waterway that have been classified as General Use and attain the aquatic life protection use.

(6) Controls more stringent that those required by Sections 301(b)(1)(A) and (B) and 306 of the Act would result in substantial and wide spread adverse social and economic impact.

While the General Use thermal standard is necessary and appropriate to protect the aquatic community otherwise attainable within the Upper Dresden Island pool, economic and operational considerations may be significant and should be given due consideration in the development of any alternate standards and the compliance period to attain that new standard. The Agency should work closely with Midwest Generation and other affected thermal sources to accurately estimate the technical, financial and scheduling requirements. If attainment of the Illinois General Use Standard is found to cause a substantial and wide spread socio - economic impact, we recommend that a new standard include a maximum temperature that represents the upper bound to prevent lethality of known indigenous fish species and additional criteria to address general growth and health needs of aquatic life effects. Figures 2.44 and 2.45 clearly document that the current General Use thermal standards provide adequate protection to the potentially indigenous aquatic species that would reside in the Dresden Island pool and should, therefore, provide the reference level for the socio-economic study. This is also required by the Water Quality Standards regulations.

Other impacts of elevated temperatures in the Dresden Island Poll will also be addressed in Chapters 3 to 6.

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CHAPTER 3

SEDIMENT QUALITY

Introduction

This chapter describes and assesses sediment contamination by pollutants. As with water quality, significant changes occurred in the Des Plaines River watershed over the last thirty years that altered the sediment quality. Sediment in the Lower Des Plaines River was and still is perceived by many as heavily contaminated and impeding the attainment of the uses of the water body that would be commensurate with the goals of the Clean Water Act. However, significant and far reaching water quality improvement took place in the watershed, especially at many treatment plants operated by the Metropolitan Water Reclamation District of Greater Chicago and other communities. These effluent improvements and building of the Tunnel and Reservoir Project (TARP) dramatically reduced the input of contaminants and contaminated solids settling in the river.

Frequent navigation also contributed to changes in sediment composition. Barge traffic constantly resuspends deposited solids that move downstream at an accelerated rate. At some sections of the river scouring by barge traffic removed most of the deposited sediment and reduced or prevented deposition. Consequently, the channel bottom in some sections is made of bedrock materials and coarse texture sediments (large sand and gravel) and not of fine contaminated sediment. In other sections, however, contaminated sediment can be still found outside of the navigational channel. Sediment composition has been studied for many years and the data provide historic information on trends in sediment contamination.

Historic Perspectives

In 1971, an extensive study of bottom sediments of the Upper Illinois River was conducted and reported by Butts (1974). The study's objective was to quantify the sediment oxygen demand (SOD) that was deemed to be an important component of the dissolved oxygen balance of the river. The study extended from Chillicotte upstream to Lockport and gathered important information on the sediment quality and its biotic status.

Butts described the sediments (in 1971) in the Brandon Road and Dresden Island pools as it *"would fit that of a thick black fibrous muck having either an oily or musty smell. The fibrous material ... was often found to be massive populations of sludge worms. ...the bottom sediments in these two pools can be characterized simply as highly infested with pollution-tolerant organisms. Heavy oils of petroleum products are widely distributed throughout most of the sediments in these two pools. Often gritty-sandy samples smelled of oil and frequently produced a rainbow effect in water. Many of the samples burst into flame in the kiln....The drying and dewatering characteristics of the sediments appeared to be similar to that of primary sludge"*.

Since the quality of sediments had improved in the downstream pools, Butts suggested that the Brandon Road and Dresden Island pools served as settling basins for sewage solids and sludge.

Butts noted the effects of navigation on sediment deposition, resuspension and the impact on the SOD. Barge traffic under certain circumstances created a violent scouring action that the samplers, along with most of the deposited sediment, was uplifted from the bottom. The resuspension of the sludge like sediments by barge traffic locally increased sediment oxygen demand (SOD) by an order of magnitude.

In 1971, the effect of invertebrates (predominantly sludge worms, most likely tubificide worms such as *Limnodrilus hoffmeisteri* that are typical for Northeastern Illinois) on the sediment quality in the Brandon Road and Dresden Island pools was overwhelming. Some samples that were collected in Dresden Island pool contained estimated invertebrate numbers as high as 100,000/m^2. Some contained solid mats of worms. These high invertebrate densities had a significant impact on the sediment oxygen demand (SOD). In addition to the benthic worms residing in the river, solid surfaces of the banks and bottom of the Brandon Road pool were covered by a thick slime layer made of organisms similar to those residing on the trickling filters.

The SOD measured by Butts (1974) in the Dresden Island pool ranged from 2.11 to 6.45 g/m^2-day for sediments composited primarily of sand and gravel, and 1.25 to 8.08 g/m^2-day for sediments containing dominantly silt and clay. These values were typical of other polluted streams in the Upper Illinois River system (Butts and Evans, 1978). The SOD range for less polluted and unpolluted rivers was between 1 and 2.5 g/m^2-day. However, Butts calculated the SOD for Brandon Road pool as ranging from 40 to 50 g/m^2-day. Such high rates (in 1971) are not typical of SODs of organic muck, they were exceedingly high. Butts (1974) and Butts et al. (1975) stated that shorelines consisting of riprap, walls of navigational locks and in shallow rocky areas downstream of Brandon and Lockport dams were covered by a dense healthy zoological matter similar to that of trickling filters and not by sludge and sediment deposits. Such biological masses can extract large amounts of oxygen from water, as they would in trickling filters. This may explain the high SOD values. Obviously, after the improvements in the treatment efficiencies of the upstream treatment plants, the river today has ceased to be an extension of the biological treatment process and is much healthier.

The sediment quality of the Lower Des Plaines River was again extensively analyzed by the USGS in its NAWQA study of the Upper Illinois River (Schmidt and Blanchard, 1997; Fitzpatrick et al., 1998; Sulivan et al., 1998). In 1987 samples of streambed sediments were collected by the USGS as a part of the NAWQA pilot study and reported by Fitzpatrick et al. in a form of pie chart maps that included arsenic, toxic metals, phosphorus, organic and inorganic carbon, and several geochemical elements. The results for some elements are presented in Table 3.1.

Sulivan et al. (1998) summarized the data on organic chemicals in the sediments analyzed between 1975 and 1990. The data sources included US Geological Survey, Illinois Environmental Protection Agency, US Army Corps of Engineers, and Metropolitan Water Reclamation District of Greater Chicago. Unfortunately, this report does not have much data on the sediment contamination of the investigated stretch of the Lower Des Plaines River.

Table 3.1 Summary of Sediment Contamination Data Ranges in the Lower Des Plaines River Measured in 1987 - Concentrations Given in mg/Kg

Contaminant	Brandon Road Pool	Dresden Island Pool
Arsenic	9.3 - 21	9.3 - 21
Barium	560 - 1500	560 - 1500
Cadmium	4.0 - 46	4.6 - 46
Copper	120 - 640	61 - 120
Lead	190 - 1700	32 - 190
Mercury	0.87 - 6.19	0.06 - 0.87
Nickel	45 - 130	28 - 45
Silver	3.0 - 29	2.0 - 3.0
Zinc	440 - 3200	120 - 440

The summary of the historical data in the Sullivan et al. document reported higher sediment concentrations of PCBs (> 205 μg/Kg) in both pools. Dieldrin was less than 1 μg/Kg.

Both IEPA and MWRDGC have continued collecting sediment data. Following the recommendation of the biological expert subcommittee for this study, only more recent data will be considered in the UAA. Nevertheless, the historical data have great comparative value for documenting the trends or improvements in sediment quality. The lead data were affected by the ban on leaded gasoline that was not fully implemented until the end of the 1980s'. However, lead from the pre-ban period may remain as a legacy pollution in sediments in the depositional areas.

Sediment Toxicity Study by Wright University - 1994 and 1995

More recent toxicological studies of sediment contamination were done by Burton (1995a) in 1994 and 1995. This study was a part of an extensive investigation commissioned by the Commonwealth Edison Company. The objective of the study was to evaluate the toxicity of the sediments in the Upper Illinois Waterways (UIW) that extended from the River Mile 322 where the Fisk Power Plant is located on the South Branch of the Chicago River, to River Mile 271.6 at the Dresden Island Lock and Dam on the Illinois River downstream of the confluence of the Des Plaines River with the Kankakee River. The objective of the study was to assess the toxicity of sediments and the extent of sediment contamination. The subsequent section is an abbreviated summary of Burtons (1995a) report. This report was also included as a chapter in the Commonwealth Edison (1996) aquatic ecology study.

The study evaluated the historic data but came to the conclusion that the extreme heterogeneity of the aquatic system of the Upper Illinois Waterway prohibited conclusive evaluation of spatial and temporal patterns with data that were classified as sporadic in nature. However, they concluded that in spite of these data inadequacies, it was apparent that *extreme chemical contamination* existed in many areas of the UIW. A study by Burton (1995b), that preceded the sediment toxicity study, included extensive sampling and measurements of the chemicals of concern. The author concluded that the study supported the chemical screening outcome of widespread system contamination from multiple chemicals.

A preceding study by Lawler, Matuskey & Skely, also commissioned by the Commonwealth Edison Co., identified the chemical contaminants of concern in the Upper Illinois Waterway (Table 3.2). It should be noted that this study summarized historic, mostly pre-TARP, data and may not reflect the present situation. The summary is pertinent to the entire UIW and all contaminants may not be of concern to the Lower Des Plaines River and its two pools.

Table 3.2 Priority Chemicals of Concerns in the Upper Illinois Waterway (historic compilation by Lawler, Matuskey, and Skely reported in Burton, 1995a)

Surface Waters[1]	Ammonia	
	Copper	
	Cyanide	
	Lead	
	Mercury	
Sediments	Ammonia	Chlordane
	Arsenic	Dieldrin
	Cadmium	DDT
	Chromium	Polycyclic Aromatic Hydrocarbons (PAHs)
	Copper	Polychlorinated Biphenyls (PCBs)
	Lead	
	Mercury	
	Nickel	
	Zinc	

The sediment study was carried out as a multi-tiered approach. In Tier One, a general survey of sediment toxicity was concluded using whole sediment exposure for 7 to 10 days. The test species included: the fathead minnow *Pimephales promelas*; the benthic invertebrate amphipod, scud, *Hyalella azteca,* and the benthic invertebrate midge, *Chirozomas tentans.* The greatest toxicity was found in sediments in the CSSC from the Cal-Sag to the Lockport Lock and Dam, in the Brandon pool and the Brandon Road Lock and Dam tailwaters.

In Tier Two, a more sensitive survey of sediment toxicity was conducted that sampled various habitats. A comparison of habitat types showed differences in toxicity between main channel, main channel borders, tributaries, tailwaters, lock and dam areas, and backwaters. Fine grained sediment

[1]Note that Chapter 2 of this UAA addressed in detail these pollutants and attainability of the water quality standards. Subsequent section of this chapter will focus on the present sediment contamination and its interference with the designated uses.

in depositional areas were more toxic. Critical fish spawning and larval areas located in the Brandon Road tailwaters and at the mouth of Jackson Creek contained acutely toxic sediments. *The main channel of the river and power plant discharge canals had sediments composited from sand, gravel and bedrock (due to higher velocities); these areas did not contain toxic sediments.*

In Tier Three, several more detailed investigations were conducted. Additional sites were sampled between the Brandon Road Lock and Dam and I-55 Bridge. The temperature profile of the Brandon Road tailwater was evaluated during hot weather conditions. The effects of specific stressors were evaluated in a series of experiments, including thermal effects, suspended solids, ammonia, metals, and polycyclic aromatic hydrocarbons (PAHs).

Thermal effects were tested by exposing test organisms *in situ. Ceriodaphnia dubia, P. Promelas, H. azteca*, and *C. tentans* were placed in chambers in the thermal plume of the Joliet Power Plant no. 29 and exposed for 48 hours. The first test was conducted in November 1994. In the first test the temperature in the plume ranged from 17 to 23°C and in the river it ranged from 15 to 17°C, respectively. This experiment partially failed because some test organisms died due to a shock caused by a sudden release of raw sewage and petroleum products from an unknown upstream source. The second experiment, conducted in August 1995, reflected more warm summer temperature conditions. Temperature in the reference station (Des Plaines River upstream) ranged from 28 to 31.5°C, the plume temperature ranged from 29.5 to 35.2 °C, and the temperature in the discharge channel ranged from 31 to 34°C. Cladocera had the highest mortality at all test stations, *Daphnia* mortality was greater in top (warmer) water (13 and 15% survival) with higher survival in the bottom (colder) water (43 and 53 % survival). *P. promelas* had the highest survival rate of 75% at the reference station and 40 to 80 % survival at test stations.

Subsequent laboratory evaluations of thermal effects was conducted with 7 day exposure of *P. promelas* and *H. azteca* at 15, 20, 25, 30, and 35 °C. The organisms were exposed in water only systems and systems containing sediments taken from above the Brandon Road Lock and Dam, containing high levels of ammonium (although not specifically stated, at the pH of water common to the Des Plains River, the ammoniacal form was less toxic ionic form NH_4^+- ammonium; the term ammonia commonly describes the unionized and far more toxic form, NH_3 that dominates at high pH). Burton (1995) concluded that for *P. promelas, site water and sediments were toxic as no survival was observed at 35°C*. However, this statement and conclusion may be incorrect since the survival of the fish was also significantly diminished in 35°C control samples (Figure 3.1). The survival of *Hyatela azteca* was also greatly reduced at 35°C at all samples and dropped to almost zero survival in water control samples that did not contain contaminants (Figure 3.2). It appears, the only reason for almost 100 % mortality was temperature[2]. Burton also observed that ammonia production

[2]Burton also made a statement that the effects observed at 35°C do not occur in the UIW because organisms are not exposed to 35°C water for 7 days or a longer period. This may not be correct today, see Figure 2.46 that indicates that temperature of 37.8°C (100 °F) might have been maintained or exceeded in 1999 in the Upper Dresden Island pool for a period of two months.

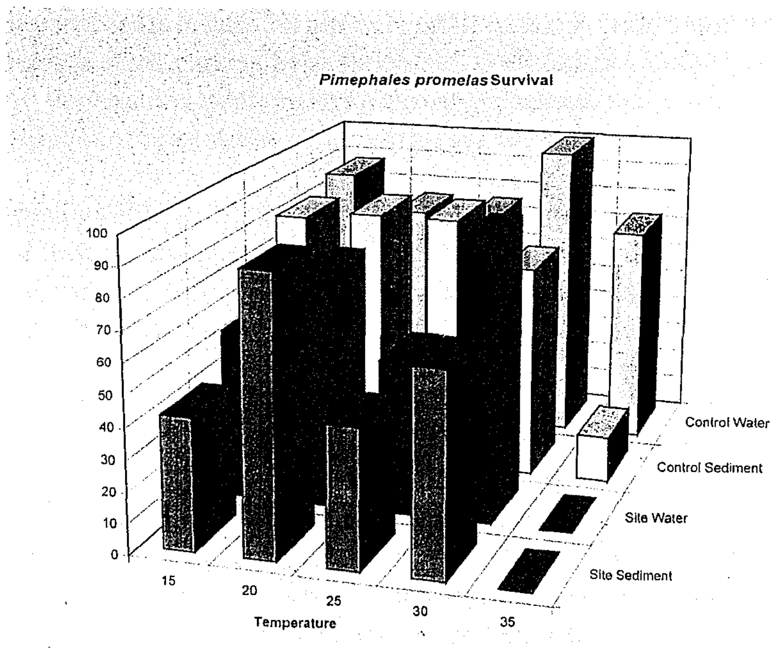

Figure 3.1 **Effect of Temperature on Survival of *P. promelas* in Burtonís (1995) Experiments with Contaminated Sediment**

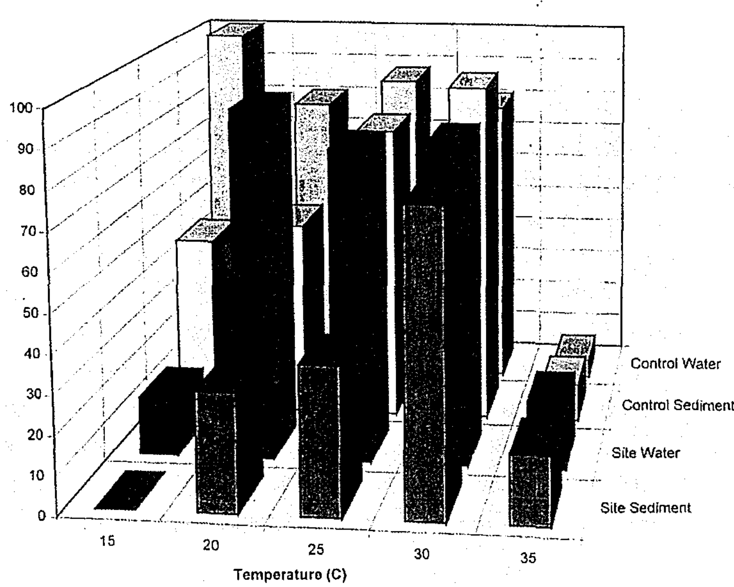

Figure 3.2 **Effect of Temperature on Survival of *H. azteca* in Burtonís (1995) Experiments with Contaminated Sediment**

increased in the sediment when temperature changed from 15 to 35°C (0.68 mg/L maximum); however, in water it only increased slightly to 0.1 mg/L. It was suggested that other stressors in the samples, e.g., metals and/or organics, increased the adverse effects of continuous exposure at 35°C and *the toxic effect appeared not to be related to ammonium since water concentrations of ammonium were very low*. *H. azteca* showed a more significant effect from sediment exposures than did *P. promelas*. In contrast to fish response, less survival was observed in sediment and site water treatment at cold temperatures, as compared to controls. Again, ammonium did not affect the survival.

Burton also studied possible effects of barge traffic and ensuing turbidity but no apparent major effects were observed for *P. promelas* and *H. azteca*. However, *C. dubia* did show some adverse effects of exposure to elevated turbidity.

Thus, Burton's experiments effectively discounted ammonium as a toxicity factor with an exception of a special experiment with the sediment taken from above and below the Brandon Road Dam. In this particular experiment, survival of *P. promelas* and *C. dubia* in unaltered sediment was 100%, with the exception of *C. dubia* survival of 75%. All organisms in the pore water only were killed[3]. Burton then exposed the sediment to ultraviolet light that released PAHs and photoactivated PAHs to more toxic form. This resulted in no survival of *C. dubia*. Metal removal from the sediment did not affect the survival rates. Burton then attributed the toxicity to ammonia, which may contradict his previous finding of no ammonium toxicity in other samples because of low ammonia levels. Burton himself classified this isolated finding as being *"uncertain since ammonia toxicity was not observed in whole sediment assays"* (p.10).

He also discounted metals as a source of toxicity: *"..metal concentrations in sediments did not appear to be a significant class of contaminants..* (p.52). As a matter of fact cadmium concentrations were positively correlated with growth of the test species, which he acknowledged to be a statistical oddity. Burton measured only the total concentrations in the sediment not their bioavailable (toxic) fractions.

Burton's (1995a and b) reports represent valuable research that provided insight and answers to the effects and extent of sediment and temperature effects on the integrity of the Lower Des Plaines River and the entire Chicago Waterways System. The conclusions drawn from the Burton's research by the AquaNova/Hey Associates team, relevant to this Lower Des Plaines River UAA, are:

- Fish (fathead minnow - *Pimenthale promelas*) after 7 days exposure did not survive in water that was 35°C (95°F). This agrees with the literature findings (e.g., Andersen, 1959) depicted on Figures 2.44 and 2.45. **The almost 100 % mortality of *H. azteca in 35 C warm control sample water can only be attributed to the high temeprature because survival with the sediment of the same quality was almost 100 % at 30 C.*** **It was pointed out in Chapter 2, this *35 C* lethal temperature is less than the Illinois Secondary Contact and**

[3] It is a known fact that sediment organic matter, sulfides and other ligants detoxify the sediment and reduce toxic levels in the pore water. See the subsequent discussion.

Indigenous Aquatic Life standard. Consequently, this standard would not provide protection against the lethal effects of temperature. The lethal effect was not related to ammonium toxicity of the sediment. Although other stressors were suggested, no proof was provided. Priority metals were not a cause of the lethality and were discounted as a source of toxicity.

- The sediments in the main channel of the river and discharge channels of the power plants were generally composited of sand, gravel and bedrock and, generally, were not toxic.

- Contaminated and potentially toxic sediments were located in depositional areas. *These toxic sediments became more patchy in their distribution in lower reaches* (of the Upper Illinois Waterway that includes also the Lower des Plaines River), *which likely reflects downstream transport and inputs of less contaminated sediments from local sources* (p.10 Burton's (1995a) report).

- The tailwater of the Brandon Road Dam contains potentially toxic sediments. The area upstream of the dam is depositional while the downstream tailwater zone receives the effluent and CSOs from the City of Joliet. The area downstream of the dam has the best habitat conditions for spawning and reproduction of fish.

- Elevated turbidity and suspended solids, due to resuspension of the sediment by barge traffic, had no impact on *P. promelas* and *H. azteca.* However, when *C. dubia* was exposed to high turbidity levels, significant mortality was observed.

- Generally, the sediment from above the Brandon Road Dam were not toxic when undisturbed. Under UV light exposure, PAHs released from the sediments became toxic and resulted in mortality of test organisms[4].

Evaluation of Toxicity of Sediments

Currently, there are no standards in force for contaminated sediments. Concentrations expressed in mg/kg or μg/kg (mass of contaminant per kg of dry weight of sediment) do not express toxicity of the contaminant and cannot be used for legal assessment of toxicity of the sediment. Many studies have shown (see USEPA, 1993; or Novotny and Olem, 1994 for reviews) that there is essentially no relationship between sediment chemical concentrations on a dry weight basis, such as that measured by the Illinois EPA and MWRDGC and biological effects, i.e., toxicity (Short, 1997). Benthic organisms are affected primarily by the dissolved concentrations of the contaminants in pore water of the sediments and not by the total mass of the contaminant in the sediment. Typically, only a fraction of a percent of the total contaminant in the sediment is dissolved in the pore water and, hence, toxic. The rest exists precipitated, adsorbed or complexed non-toxic forms.

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[4] This would be an unlikely scenario in -situ because these sediments are under more than 9 ft of water.

In Chapter 2 we have discussed a possibility of developing site specific sediment quality criteria for copper in the Lower Des Plaines River. We have assembled information from the US Environmental Protection Agency (Great Lakes Environmental Center, 2001) on the toxicity of copper to various benthic organisms that reside or could potentially reside in the benthos of the Lower Des Plaines River. However, we did not advance this idea any further because of the high uncertainty of the magnitude of the partition coefficient and absence of measurements of acid volatile sulfides in the sediments that would allow a more precise determination. One interesting deduction can be made from the Butts (1974) observation of the invertebrate composition of the benthic layer and their extremely high densities in the Brandon Road and Dresden Island pools. It was stated in the preceding section that in the early 1970s, the bottom in many sections contained an abundance of sludge worms, most likely *Limnodrilus hoffmeisteri* or *Tubifex tubifex*. In several locations on the Brandon and Dresden Island pool, these worms were the only invertebrates found in significant numbers. The sludge worms are highly resistant to organic pollution, as a matter of fact they thrive on it. Tubificid worms borrow into the upper layer of the sediment and derive their food from the sediment (Figure 3.3). However, they are sensitive to toxic contamination, especially in pore water of the sediments. Their LC(50) for toxicity of copper and other metals is less than that for sensitive fish (e.g., salmon and trout) and only marginally greater than that for daphnia, the most sensitive species. For example, copper LC(50) for Tubifex (ranked No 14) converted to hardness of 50 mg $CaCO_3$/L is about 35 µg/L, that for *Limnodrilus hoffmeisteri* (ranked 19), is about 50 µg/L, and for Daphnia magna (ranked No 2 in sensitivity, not a benthic organism) is about 13 µg/L. In contrast, LC(50) of copper for Brook Trout, one of the most sensitive fish ranked No 42, is 110 µg/L (Great Lakes Environmental Center, 2001). It must be emphasized that these concentrations apply to pore water and not to total concentrations in the sediment. Also hardness of pore water in sediments of the Des Plaines River is greater than 50 mg $CaCO_3$ for which the LC(50) toxicity levels were defined in the toxicity report (Great Lakes Environmental Center, 2001). This would imply that in 1971 the sediment, in spite of its high pollution, most likely was not toxic to fish and many other organisms. The organic matter, clay and sulfide content may have detoxified the potentially toxic contaminants contained in the sediment.

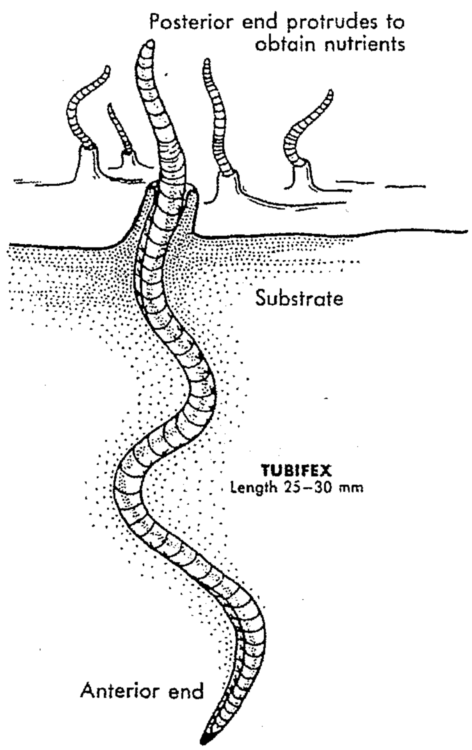


Figure 3.3 Tubificid Worms Derive Their Nourishment from the Sediment. These Organisms Are Tolerant of High Organic Pollution but Intolerant of Toxic Metals.

The capacity of sediments to adsorb, retain and essentially detoxify contaminants depends on their composition. For organic micro pollutants, the most important detoxifying component in soils is the organic particulate matter which has the strongest binding capacity. This is one of the reasons why the organic matter content of sediments should be considered when defining soil and sediment pollution standards and their toxicity such as it was done, for example, in the Netherlands or proposed to the USEPA by a sediment toxicity task force (DiToro et al., 1991 a and b). For inorganic contaminants, such as toxic metals the adsorbing capacity of both organic and inorganic soil/sediment particulates should be considered. The adsorbing capacity is related to the surface area of the particles. Hence, small particles like clay minerals have the highest adsorbing capacity. Salomons and Stol (1995) identified the parameters that control the capacity to retain organic toxic pollutants (Capacity Controlling Parameters- CCP). The important CCPs are the soil organic matter content, redox status and the sum of "cation exchange capacity," which is determined by the surface area and nature of the particles.

Toxic Metals - Complexation and Immobilization in Sediments

When metals are added to water and settle into sediments they undergo complexation with *ligands* that can be both inorganic and organic. Because the metals exist in aqueous solution as positively charged cations, ligands are mostly negatively charged anions that bond to the metal ion. Examples of inorganic ligands include OH^-, SO_4^{2-}, CO_3^{2-}, Cl^-, S^{2-}, PO_4^{3-}, NO_3^-, and others. Organic ligands are humic substances that form from the decomposition of vegetation (Fetter, 1999). Complexation is important because the free metallic ions (for example, divalent toxic metal ions such as Cd^{++}, Cu^{++}, Pb^{++}, Zn^{++}) or methyl-metal complexes are far more toxic than other less soluble complexes. Many metal complexes are not biologically available and, hence, are not toxic.

Major causes for precipitation of metals, metalloids and metal complexation are (Salomons and Förstner, 1984):

1 Oxidation of reduced components such as iron, manganese and sulfides
2. Reduction of higher valency metals by interaction with organic matter (selenium, silver)
3. Reduction of sulfate to sulfide (iron, copper, silver, zinc, mercury, nickel, arsenic, and selenium are precipitated as metal sulfides) that occurs in anaerobic sediments.
4. Alkaline-type reactions (strontium, manganese, iron, zinc, cadmium, and other elements are precipitated by increased pH, usually caused by interactions with alkaline rocks and sediments or by mixing with alkaline waters)
5. Adsorption or co-precipitation of metallic ions with iron and manganese oxides, clays, and particulate organic matter in aerobic sediments and soils.
6. Ion-exchange reactions, primarily with clays and, to a lesser degree by Fe and Mn oxides.

Complexation and precipitation processes for metals are pH dependent. Jørgensen (1995) listed several examples of pH effects:

1. Solubility and, consequently, the release of metals from sediments and soil increases with decreasing pH. Concentration of sulfide ions decreases with decreasing pH, as sulfide ions react with H^+ and form hydrogen sulfide.

2. Most ligands are acid-base systems and; therefore, have different forms at different pH values.
3. Hydroxides of toxic metals have very small solubility products, most metals will precipitate at pH >7.5.
4. Many metals react with water by formation of metal-hydroxides and hydrogen ions.
5. Toxic substances are able to form a number of species as a result of hydrolysis.

Metal solubility is also greatly affected by oxidation-reduction conditions. In aerobic freshwater sediments the sorption sites are provided by organic carbon, clays, and hydrous oxides of iron and manganese. The Fe and Mn oxides also have limited ion exchange capabilities. Hydrous iron oxides strongly adsorb chromium, while manganese oxides adsorb nickel, and calcium phosphate (also present in sediments) adsorbs cadmium, lead, and other metals. Mercury in sediments (in sediments mercury exists mostly as methyl mercury) is strongly adsorbed by organic matter (Langston, 1985). Oxides of iron and manganese are deemed to be more important than organic matter and clays; however, Combest (1991) documented that the Fe and Mn contents correlate with the clay content.

In anaerobic sediments and soils, iron and manganese are reduced and sulfide precipitation becomes important for complexation of toxic metals (DiToro, 2000; DiToro et al., 1989; Jørgensen, 1995; DiToro and DeRosa, 1995; Salomons, 1995). Therefore, sulfides become the most important ligands. Metal-sulfide complexes are insoluble and biologically unavailable.

In summary, the adsorbing and complexing compounds for toxic metals include:

1. Particulates: sulfides, iron and manganese oxyhydrates, particulate organic matter, clays
2. Dissolved: sulfides, humic compounds, organic acids, hydroxyl ions

The free metal ion is the most toxic component for organisms (Salomons and Förstner, 1984; DiTiro and DeRosa, 1995; Jørgensen, 1995; Novotny and Olem, 1994). When metal ions are present in water they are distributed with the various complexing ligands and solids.

DiToro et al. (1989) and DiToro and DeRosa (1995) reported that in sediments the concentration of metal-ligand complexes in pore water is negligible when compared to that adsorbed on the sediment or soil particles. Then, neglecting the ligand-metal concentrations in pore water, the pore water free metal concentration in aerobic sediments and soils becomes

$$M^{2+} = \frac{M_s}{K_{Fe}[FeO_x] + K_{Mn}[MnO_x] + m_{ss}(K_{oc}\,f_{oc} + K_{clay}\,f_{clay})}$$

Hence, the denominator of the above equation could be called a "partition coefficient" for metals or

$$\Pi = K_{Fe}[FeOH_x] + K_{Mn}[MnO_x] + m_{ss}(K_{oc}\,f_{oc} + K_{clay}f_{caly})$$

and similarly to the partitioning relationship introduced previously in Chapter 2 (for analysis of sediment contamination by copper)

$$M_s = \Pi M^{2+}$$

In anaerobic sediments and saturated soils, iron-manganese oxyhydrates are reduced and sulfides are the dominant ligand. Therefore, $K_{S(2-)}[S^{2-}]$ replace oxide terms in the partitioning equation above.

The pH effect on the adsorption and complexation reactions is very strong, ranging from zero adsorption in low pH to 100 % adsorption/precipitation in higher pH. Speciation of metals can be estimated and/or simulated by the USEPA model MINTEQA2 (Allison, Brown, and Novo-Gradac, 1990).

The above discussion indicates that evaluation of toxicity of sediments for metals is complex. In Chapter 2 we have presented a simplified analysis of partitioning of copper in water and sediments. It was found that for the conditions of the Lower Des Plaines River, the pore water concentrations could exceed chronic toxicity of copper in water and could be classified as mildly contaminated. However, upon resuspension, because the partition coefficient for copper in sediments is smaller than that for water, the sediment would scavenge copper from water and actually reduce the metal content of water and eventually resettle into the benthic layer. This may be true for other metals.

Organic Toxic Chemicals

Generally, water-soluble (hydrophilic) organic compounds are weakly adsorbed on sediment particles. Water-insoluble compounds (hydrophobic), on the other hand, are immobile in sediments; however, they accumulate in sediments and may bioaccumulate in organisms and biomagnify in the food chain.

The mobility of an organic chemical (micro-pollutant) in sediments is related to the *octanol-water partitioning coefficient,* K_{ow}. This coefficient is correlated to the solubility of the compound in water and to the controlling parameters. Consequently, K_{ow} is a measure of mobility of the pollutant in soils and sediments. The values of K_{ow} for some very environmentally important chemicals (priority pollutants) were summarized in Schnoor et al. (1987), Novotny and Olem (1994) and Ambrose (1999). Concepts were explained in detail in DiToro (2000), and Schnoor (1996).

The relationship of dissolved (pore water) concentration of a chemical and its total concentration in the sediment is

$$c_d(\mu g / L) = \frac{1}{\theta + \Pi m_{ss}} c_T(\mu g / L) \qquad \text{or} \qquad C_d(\mu g/L) \approx C_T(\mu g/Kg)/\Pi$$

where Π is the partition coefficient related to sediment organic matter and K_{ow}. The organic chemicals of concern identified for the Des Plaines River in the 305(b) report in the sediments of the Des Plaines River are Polyaromatic Hydrocarbons (PAHs) and Polychlorinated Biphenyls (PCBs).

PAHs found in sediments originate generally from diffuse sources such as automobile and truck traffic, municipal and industrial wastewater effluents (point sources), forest fires, and combustion and gasification of coal. Automobiles, especially those with diesel engines, were in the past a major source of PAHs. Recent restrictions on emissions have significantly reduced their discharge.

Typically, urban runoff contains measurable quantities of PAHs that are mainly incorporated into sediments. Sediment microorganisms are capable of degrading PAHs. Photolysis is also an important degradation process for some PAHs (for example, anthracene); however, Burton's experiments showed that photolysis (exposure of sediments to ultraviolet light) may also work in the opposite direction and make the sediment more toxic. PAH will be addressed in more detail in the subsequent section.

Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) are man-made chemicals that are alien to nature and, as with most of the human produced organic chemicals (rare exceptions are some PAHs), no natural-background concentrations in soils and sediments exist. Most of the environmental mass of PCBs is confined to industrial and urban areas; however, PCB contamination is global and PCB has been measured in polar glaciers. Many freshwater and aquatic sediments have been heavily contaminated by these compounds (e.g., Waukegan, Illinois, Harbor, ponds on Cedar Creek in Cedarburg, Wisconsin, and Green Bay on the Lake Michigan are examples of such environmental damage). They have been found in the sediments of the Lower Des Plaines River (Burton, 1995a and b, see also the subsequent section on the USEPA 2001 survey). The sources of these contaminations were traced to past industrial operations such as past discharges of cooling liquids in tool and dye manufacturing, transformers liquids, and paper production (Novotny and Chesters, 1981) such as in the Fox and Sheboygan Rivers in Wisconsin.

PCBs have very low solubility, consequently, their octanol partition coefficients are large; typically, K_{ow}, would range between 10^4 and 10^6 L/kg. PCBs are difficult to decompose in the sediments and their persistence is related to the number of chlorinated sites in the two ring molecule. The compounds that have a larger number of chlorinated sites are most persistent. The removal of PCBs from soils is primarily by volatilization and biomodification of lower PCBs.

Ammonium

Much has been said about the potential toxicity of the Des Plaines River sediments caused by ammonium.. In the absence of high concentrations of ammonium in water documented in Chapter 2, ammonium in sediment develops from the breakdown of the sediment organic matter. This organic matter may be of natural as well as anthropogenic origin. The sources may be algal development and settling and sewage solids from CSOs. Both contain organic nitrogen. The process of ammonium diagenesis along with the formation of methane under anaerobic conditions in riverine sediments was eloquently described by DiToro et al. (1990) and DiToro (2000), based on observations and model development of the sediment oxygen demand for the Milwaukee (WI) Inner Harbor.

The *diagenesis model* proposed by DiToro and co-workers and shown on Figure 3.4, relates SOD to the input of particulate organic matter into the bottom sediment layer and its anaerobic decomposition. The process in organic sediments is similar but not identical to the anaerobic processes of breakdown of organic particulate matter in sludge digesters or decomposition occurring in wetlands. In this process, reduced soluble species - CH_4(aq.)), HS^-, Fe^{2+} and NH_4^+ - are produced. These soluble compounds (part of methane may be in a gaseous form and escapes from the sediment

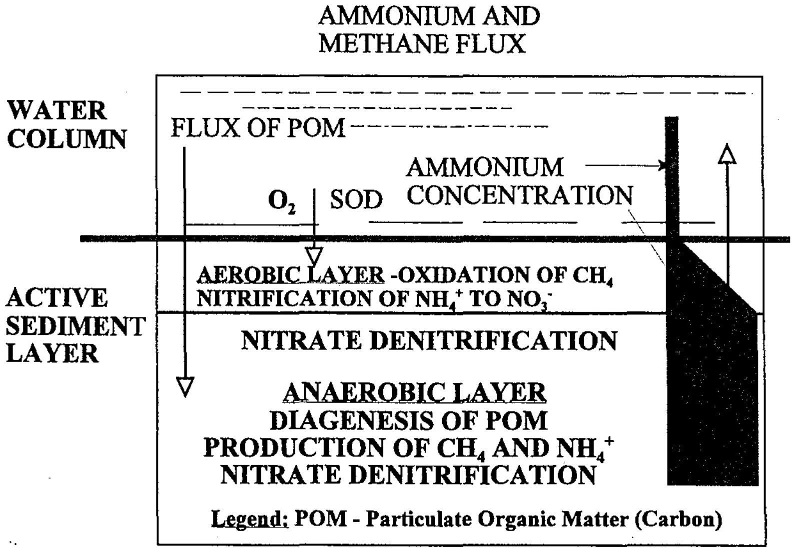


Figure 3.4 Concept of diagenesis in sediments (after DiToro et al., 1990)

as bubbles) move due to the concentration gradient by diffusion towards the interstitial layer between the sediment and water. The production of the soluble reduced end products in the sediments occurs via the bacterial breakdown of particulate organic matter. The most important products of the breakdown of organic matter are carbon dioxide, methane and ammonium/ammonia.

The interstitial aerobic layer (Figure 3.4) on top of the sediment is rich with microorganisms. If the overlaying water has oxygen then the interstitial layer is aerobic, which may occur today in the Brandon and Dresden Island pools of the Lower Des Plaines River. Both heterotrophic bacteria oxidizing methane and nitrifiers oxidizing ammonium reside in this layer. DiToro then has proven mathematically that the SOD is equivalent to the oxygen demand of the reduced species (electron donors) such as organic carbon (CH_4(aq.)), HS^-, Fe^{2+} and NH_4^+. This may explain the elevated SOD of the sediments measured previously by Butts and his coworkers. If the sediment layer had been highly toxic (to bacteria), no or only small SOD would have been measured. A mass balance equation of the oxygen demand equivalents is used to calculate their flux in the sediment water interface, a consequence of which is the SOD.

At lower temperatures (up to 25°C) almost all ammonium is oxidized in the upper aerobic layer of sediment. The produced nitrate, again due to the concentration gradient, moves by diffusion back into the sediment (not to water because the water is rich with nitrate) where it is converted by facultative bacteria in the anoxic sediment to nitrogen gas that escapes. This process is called *simultaneous nitrification/de-nitrification* that has been recognized as a common nitrogen sink (Keeney, 1973; Keeney et al., 1975). However, nitrification rate is progressively reduced at temperatures above 22°C and at 35°C nitrification progresses at a rate of about 50% of its optimum at 22°C (Zanoni, 1968) while diagenesis of ammonium from decomposition of organic matter in sediments progresses at an accelerated rate. This suppression of nitrification at higher temperatures may explain the ammonium toxicity problem in Burton's experiments. Nevertheless, based on the diagenesis concept and presence of the surface aerobic sediment layer in the Des Plaines River (because there is enough oxygen in the overlying water) it is unlikely that aerobic benthic organisms (bottom feeding fish, mussels, worms) will be adversely affected by ammonium that is below the surface benthic layer. It is more likely that the concentrations of the ammonium in the upper sediment layer may be more close to the ammonium concentrations in the overlaying water than to that measured in the anaerobic sediment below the superficial aerated sediment layer. Most of the benthic organisms reside in the upper layer. The data on ammonium concentrations should be compared to the values of the Total Kjeldahl Nitrogen (TKN), which is a sum of ammonium and organic nitrogen. If TKN is high and NH_4^+-N is relatively

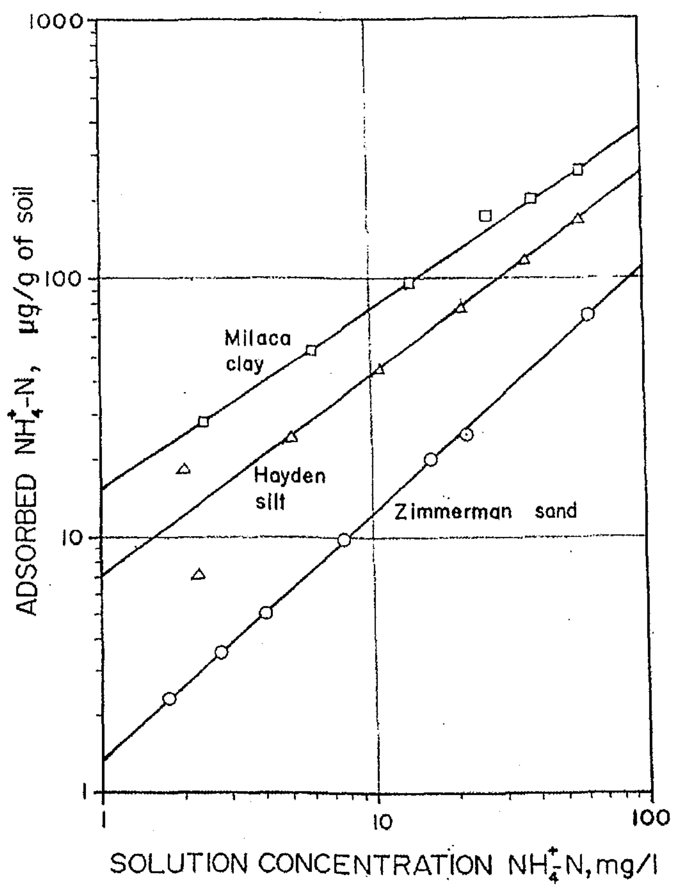


Figure 3.5 Adsorption Isotherm of Ammonium on Soils (from Preul and Schoepfer, 1968)

low, then it is likely that the ammonium is a byproduct of the sediment diagenesis, which may be natural. The sediments with elevated ammonium content are located in the depositional areas where releasing ammonium by resuspension by barge traffic is unlikely. As documented by Burton (1995a), sediments in the navigational and heated discharge channels are not toxic.

In addition to the conversion of ammonium to nitrate, ammonium in sediment can be partially detoxified (immobilized) by adsorption on the sediment clay and particulate organic matter (Preul and Schoepfer, 1968). The adsorption isotherm for ammonium in soil and sediment is in Figure 3.5. The isotherm relates the ammonium adsorbed on the sediment to the concentration of the ammonium in the pore water of the sediment. The total ammonium concentration on the sediment is then the sum of the two fractions. Thus, if the total concentration of the ammonium in the sediment is about 80 mg/Kg of the dry weight of the sediment composited of a mixture of clay and silt particles, the pore water concentration would be about 12 mg/L, which, as documented in Chapter 2 may not be toxic based on the current water quality standards for ammonium toxicity to aquatic organisms. However, as pointed out previously in Chapter 2, chronic toxicity of ammonium/ammonia is related to temperature and high temperatures increase the toxicity. It should be noted that the pore water concentrations of ammonium in Burton's (1995a) experiments ranged from 0.4 to 6.4 mg/L, which is well below toxic levels. Burton himself classified the ammonium toxicity of the sediments used in his experiments as low to moderate and well below the acute toxicity thresholds (24 to 60 mg/L in pore water) for the two sensitive species used in the experiments, *Hyadella azteca* and *Ceriodaphnia dubia* (p.42).

This discussion of the ammonium toxicity in sediment by no means tries to downgrade the concerns about the toxicity ofthe sediments and ammonium in particular. However, stressors or a combination of stressors other than ammonium may be responsible for the low biotic integrity of the Brandon Road and Dresden Island pool in the sediments of the Burton's experiments.

Comparative Criteria for Sediments and Sediment Contamination

In the absence of a quantitative measure (standard) for sediments, the Illinois EPA statistically ranked the contaminated sediments in the state's surface waters (Short, 1997). Based on the ranking, a sediment classification scheme was developed. This scheme classifies the sediments into quasi-arbitrary categories of noneleveated, elevated and highly elevated sediment contamination (Table 3.3). This simple classification was justified by the desire of the Illinois EPA to find out where are the streams with nonelevated, elevated and highly elevated sediment contamination and what are the 85 and 98 percentiles of contamination. These percentiles were selected to correspond to one and two standard deviations above the mean concentration of the Illinois sediments. This classification does not provide answers to the question of whether the sediments are toxic or nontoxic to benthic biota. This categorization follows the one used by the Illinois EPA for lake sediment classification.

Measurements of the Sediment Quality by the MWRDGC 1983 - 2000

MWRDGC has been conducting sediment quality monitoring since 1983. This data base provides information on more recent sediment quality and the trends over the last twenty years. Figures 3.6 to 3.8 show historical comparisons of sediment contamination. It can be seen that significant improvement in sediment quality have been achieved in the last 13 years. This supports the finding and recommendation of the biological experts subcommittee that only the last five years of data may provide reliable information on the current status of the sediment quality. It should be noted that some parameters included in Table 3.3 cannot be characterized as pollution, at least not in the same category as toxic priority pollutants. For example, COD and Volatile Residue are measures of organic content of sediment but not of pollution. The same characterization also applies to iron and manganese.

Members of the biological expert subcommittee also objected to developing standards for the sediment contamination using the sediment - pore water partitioning concept described in the preceding section. This apprehension may be justified because the available information that is needed for such calculations is incomplete or nonexistent. For example, information on the volatile sulfide content of sediment, the key parameter for calculation of partitioning of metals in aquatic sediments, was not available. General magnitudes of the partition coefficients reported in literature vary by orders of magnitude. In an analysis of the toxic impact, the affected organisms would have to be identified and a criterion would be developed based on the benthic and bottom feeding representative species in a process similar to that outlined in Chapter 2 for development of site specific criteria for cooper. This would require a specific focused study that would have gone beyond the scope of this UAA. The sediments of the Lower Des Plaines River are constantly being resuspended and moved downstream by barges. Therefore, the quality of the sediment is constantly changing and, it could be said, improving. This is illustrated on Figures 3.6 to 3.8 for four metals.

Tables 3.4 to 3.6 summarize the MWRDGC monitoring data for the sediments in the Brandon Road and Dresden Island pools for the years 1987 to 1989, 1994 to 1995 and 1999-2000. Values measured by Burton (1995) in the areas close (but not identical) to the MWRDGC sites were added for comparison. In the tables, the values for the 1999-2000 period were compared with the comparative

Table 3.3 Provisional Classification of Illinois EPA Sieved Stream Sediment Data Based on Percentiles (In Sediment Dry Weights)*

Classification parameter	Concentration	Nonelevated <85%	Elevated ≥85%	Highly elevated ≥98%
Phosphorus	mg/kg	<1000	≥1000	≥2800
Kjeldahl Nitrogen	mg/Kg	<2950	≥2950	≥4680
%Volatile residue	%	<8.4	≥8.5	≥13
Arsenic	mg/kg	<7.2	≥7.2	≥18
Barium	mg/kg	<145	≥145	≥230
Cadmium	mg/kg	<2.0	≥2.0	≥9.3
COD	mg/kg	<77 800	≥77 800	≥150 000
Chromium	mg/kg	<37	≥37	≥110
Copper	mg/kg	<37	≥37	≥170
Lead	mg/kg	<60	≥60	≥245
Mercury	mg/kg	<0.28	≥0.28	≥1.40
Nickel	mg/kg	<26	≥26	≥45
Silver	mg/kg	<5	na	>5
Zinc	mg/kg	<170	≥170	≥760
PCBs	μg/kg	<10	≥10	≥480
Aldrin	μg/kg	<1.0	Na	≥1.0
Dieldrin	μg/kg	<1.0	≥1.0	≥15
DDT Sum	μg/kg	<1.0	≥1.0	≥110
Total Chlordane	μg/kg	<5/0	≥5.0	≥38
Endrin	μg/kg	<1.0	Na	≥1.0
Methoxychloor	μg/kg	<5.0	Na	≥5.0
Alpha BHC	μg/kg	<1.0	Na	≥1.0
Gamma BHC	μg/kg	<1.0	Na	≥1.0
Hexachlorobenzene	μg/kg	<1.0	Na	≥1.0
Heptachlor	μg/kg	<1.0	Na	≥1.0
Heptachlor epoxide	μg/kg	<1.0	≥1.0	≥3.8

* From Short (1997) Na - not available

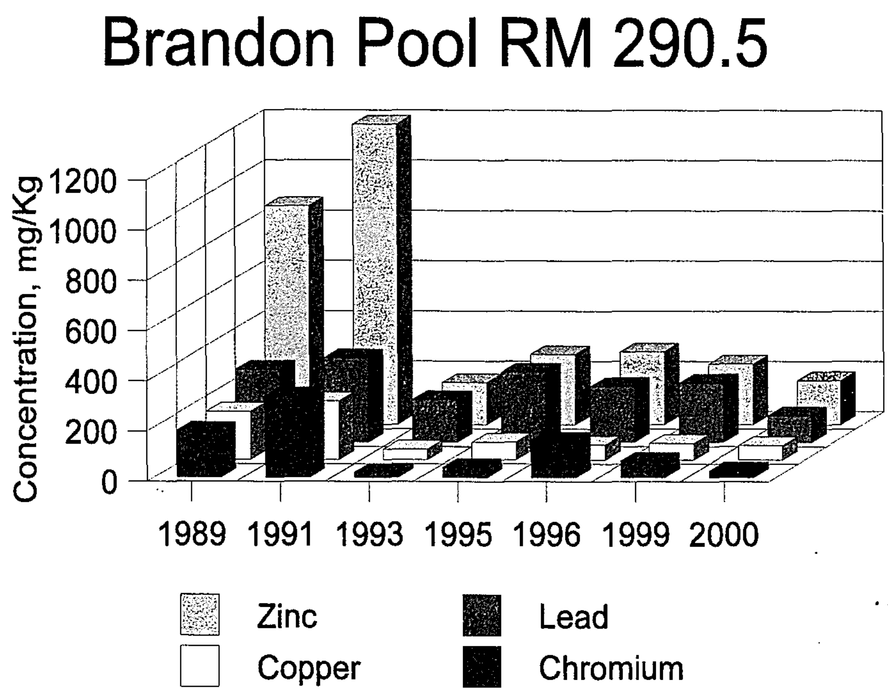


Figure 3.6 **Chronology of Sediment Concentrations of Four Metals in Brandon Pool Measured by the MWRDGC**

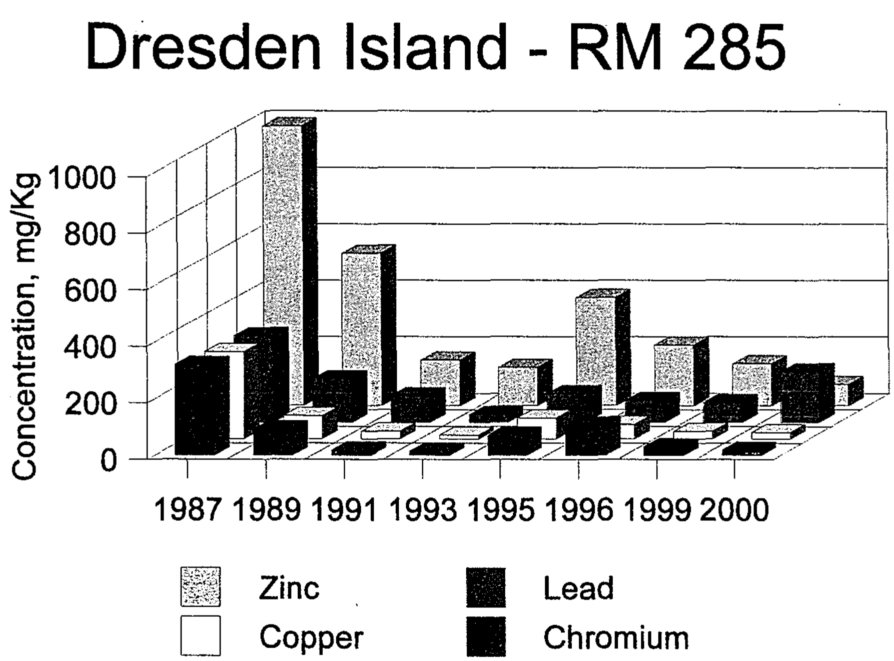


Figure 3.7 **Chronology of Sediment Concentrations of Four Metals in Dresden Island Pool Measured by the MWRDGC**

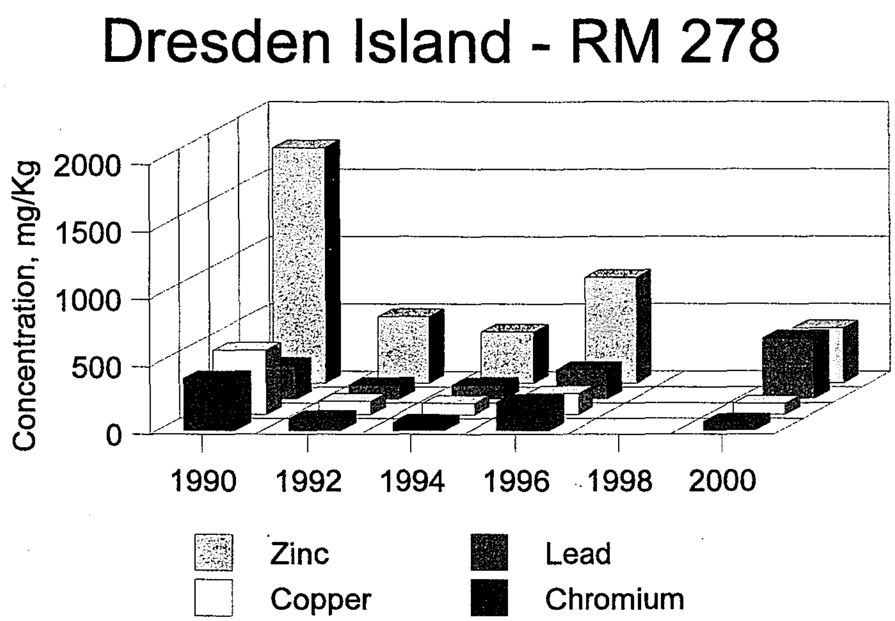


Figure 3.8 Chronology of Sediment Concentrations of Four Metals in Dresden Island Pool Measured by the MWRDGC

IEPA criteria from the report by Short (1997). The values that were *elevated* are in **bold** digits. There were no *highly elevated concentrations* reported in the MWRDGC sediment samples in 1999-2000. However, arsenic(As) that in 1996 and before was below the detection limit (<1 mg/Kg) in all sediment samples; in 1999 it reached levels at all three locations approaching the *highly elevated* values. One may speculate that an upstream As spill occurred between October 1996 and 1999. The arsenic effect on toxicity would not show in Burton's experiments because in 1994-1995 As was very low. It may be worthwhile to try locating the source (by tracing the sediment contamination).

Tables 3.4 to 3.6 and Figure 3.6 to 3.8 clearly show that there is a difference between the current sediment quality and that ten to twelve years ago. Some values measured by the Metropolitan Water Reclamation District of Greater Chicago today are less than one half of the concentrations measured in the 1980s. Also some concentrations of the sediment contaminants measured in Burton's samples were significantly higher than those measured by the MWRDGC in the nearby locations at the same time. The difference may the selection of Burton's sediments for the experiment. He used sediment collected at River Mile 286+ from the Brandon Road Dam upstream tailwater while MWRDGC collects sediments at Joliet MWRDGC 93 station at River Mile 290.5. The tailwater of the dam is a depositional area and the MWRDGC 93 location is a navigational channel with minimum or no deposition. Therefore, Burton's experiments and conclusions may not reflect current conditions of sediments throughout the Lower Des Plaines River. It will be documented in the next section of this chapter, the Brandon Road Dam tailwater sediment represents the worst case. The MWRDGC data represent an invaluable historic progression of the quality of the sediments in the Des Plaines River.

Table 3.4 Sediment concentrations of pollutants at RM 290.5 -Brandon Road Pool - CSSC downstream of Lockport

Years	TVS	TKN	NH_4^+-N	As	Cd	Cr	Cu	Pb	Hg	N i	Ag	Zn
	%	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg
1987-1989	11.7	3,605	51	<1	17	185	192	290	1.0	80	3	870
1994-1995	6.0	1,783	8	<1	4	145	60	223	0.5	39	4	290
Burton (1995)			5 - 25	8-20	23 - 27	323	100-400	300-500	1.3-3.0	100-300		>3000
1999-2000	**9.4**	1,973	14	**14**	2.5	**45**	**61**	**171**	0.3	**32**	2	**210**

Table 3.5 Sediment concentrations of pollutants in the Dresden Island Pool - Brandon Rd. Dam Tailwater - RM 285

Years	TVS	TKN	NH_4^+-N	As	Cd	Cr	Cu	Pb	Hg	N i	Ag	Zn
	%	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg
1987-1989	8.1	1,650	30	<1	5	193	42	76	0.2	46	4	566
1994-1995	5.2	453	6	<1	4	53	60	119	0.2	38	6	291
Burton (1995)			22 - 26	13								
1999-2000	4.8	648	4.5	**11**	1	25	25	**125**	0.3	25	3.5	112

Table 3.6 Sediment concentrations of pollutants in the Dresden Island Pool near I-55 - RM 278

Years	TVS	TKN	NH_4^+-N	As	Cd	Cr	Cu	Pb	Hg	N i	Ag	Zn
	%	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg
1987-1989	8.7	1,688	175	<1	18	20	175	27	2.4	67	1	680
1994-1995	4.4	3,399	24	<1	10	130	120	88	0.15	43	5	389
Burton (1995)			5 - 10	4 - 10	1 - 12	90 - 400	90 - 400	80 - 400	0.3-1.0	90-300		100-1000
99-2000	8.0	**3,263**	65	**15.5**	**4**	**53**	**63**	**94**	**0.6**	**35**	4.5	**364**

USEPA Comprehensive Sediment Survey in 2001

The US Environmental Protection Agency has conducted three detailed sediment surveys and analyses in 2001. The sampling point locations were both in the navigational channel and in the depositional areas outside of navigation traffic. Figure 3.9 shows the location of the sampling points.

The surveys were conducted in May, September and October. The May sampling used a core sampler that collect stratified sediment over a depth. The September sample collection used a ponar sampling method that scrapes sediment from the sediment surface layer about 10 cm thick. The ponar sampler composites the sediment. In October, both ponar and core sampling were used.

The sample locations were recorded in latitude/longitude coordinates and had to be converted into river miles. Also in our evaluation we separated channel and outside the channel data. The ponar data and the surface layer information of core samples is important because the benthic organisms reside either in or on the surface layer. The core data becomes important when considerations may be given to sediment remediation such as dredging.

The USEPA sampled for many pollutants, many listed as priority pollutants. Included in this report analysis are those pollutants that are included on the priority pollutant list and have defined CMC (acute) and CCC (chronic) water quality criteria.

Methods of Analysis

The priority organic pollutant content of the USEPA monitoring, with exception of metals, could not be compared with the IEPA's comparative scale. However, the data contained all parameters that could be used for relatively accurate calculation of pore water concentrations. The pore water concentrations can then be compared with water only chronic (CCC) and acute (CMC) criteria to give an approximate assessment of the toxicity of the sediment.

Following the methodology outlined by DiToro et al. (1991a) and also summarized in Novotny and Olem (1994), sediment toxicity can be expressed in terms of the sediment toxicity unit (STU) which is a ratio of the pore water (dissolved) contaminant concentration divided by the water only toxicity criterion. DiToro et al. suggested to use the chronic criterion. The pore water concentration is calculated from the total contaminant concentration in the sediment using the following well known simple equation

$$C_d\ (\mu g/L) = C_T\ (\mu g/Kg)/\Pi$$

where C_d is the pore water concentration of the contaminant, C_T is the total concentration of the contaminant in the sediment and Π is the partition coefficient. The partition coefficient for organic hydrophobic chemicals is related primarily to sediment organic matter and the octanol partitioning coefficient, K_{ow}. Both parameters can be reliably measured and are known. This makes the calculation of the pore water concentrations for organic pollutants more accurate than that for toxic metals where the key parameters for the sediment, e.g., hardness in the sediment and the sulfide content were not measured and only crude estimates of the magnitude of the overall partition coefficient obtained from

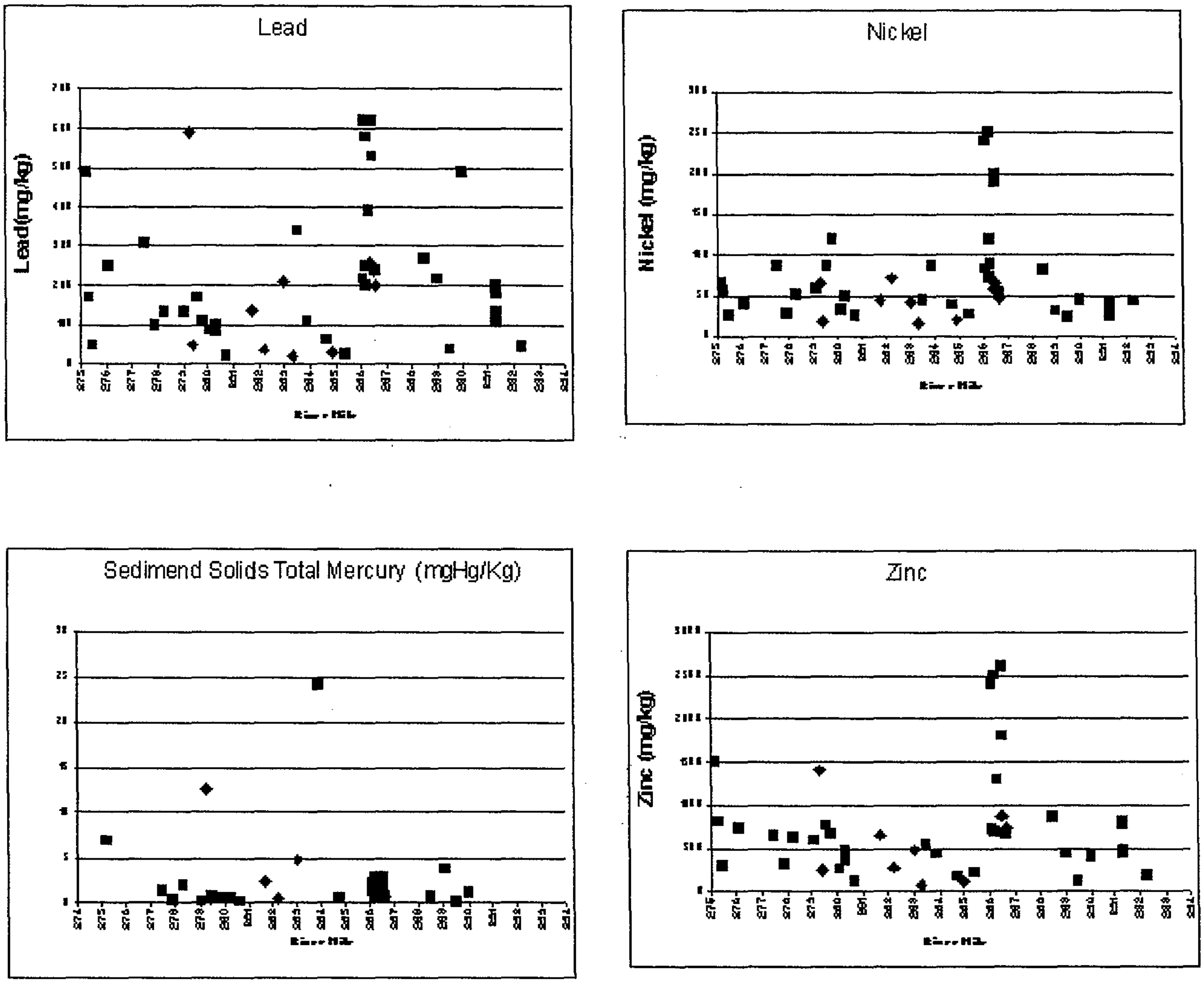


Figure 3.10 Continuing

Calculation of the pore water concentrations and the sediment toxicity unit revealed that metals do not present a toxicity problem in the river sediments with exception of cadmium at the RM 286+ depositional zone above the Brandon Road Dam (STU = 11.3). This was confirmed by Burtonís experiments and follows the finding of compliance with water quality standards in the overlying water.

Pesticides

Table 3.8 contains pore water calculations for the pesticides that have an established water quality criterion and/or standard. Calculation of the partitioning coefficient requires knowledge of the organic carbon fraction that was taken from USEPA data. The measurements of the organic carbon content of the sediment are shown on Figure 3.11. For the calculation the fraction of organic carbon was selected as

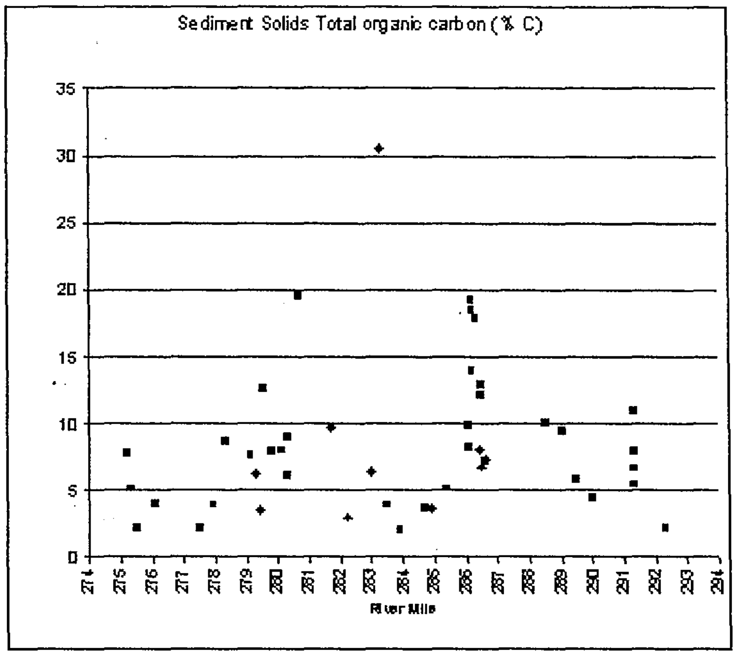


Figure 3.11 Organic Carbon Content of Sediments

$f_{oc} = 0.05$ for the river sediments, and

$f_{oc} = 0.15$ for sediments at River Mile 286+

The partition coefficient then becomes

$$\Pi = 0.63\, f_{oc}\, K_{ow}$$

Table 3.8 Calculation of Pore Water Concentrations of Pesticides

Compound	CMC µg/L	CCC µg/L	Log K_{ow}	K_{ow} L/Kg	Π L/Kg	C_T µg/Kg	C_d µg/L	STU
Aldrin	3	-	5.11	1.28×10^5	4,057	7.5	0.0018	<1
Dieldrin	2.5	0.0019	4.09	1.23×10^4	387	7.5	0.019	**10**
Endrin	0.18	0.0023	5.6	3.98×10^5	12,540	7.0	0.0006	0.26
Endosulfan	0.22	0.056	3.6	3,981	125	5.0	0.04	0.71
DDT	1.1	0.001	6.19	1.54×10^6	48,787	20.0	0.0004	0.4
Heptachlor	0.52	0.0038	4.41	25,704	810	5.0	0.0062	**1.62**
Heptachlor epoxide	0.52	0.0038	2.65	447	14.07	10	0.71	**25.4**

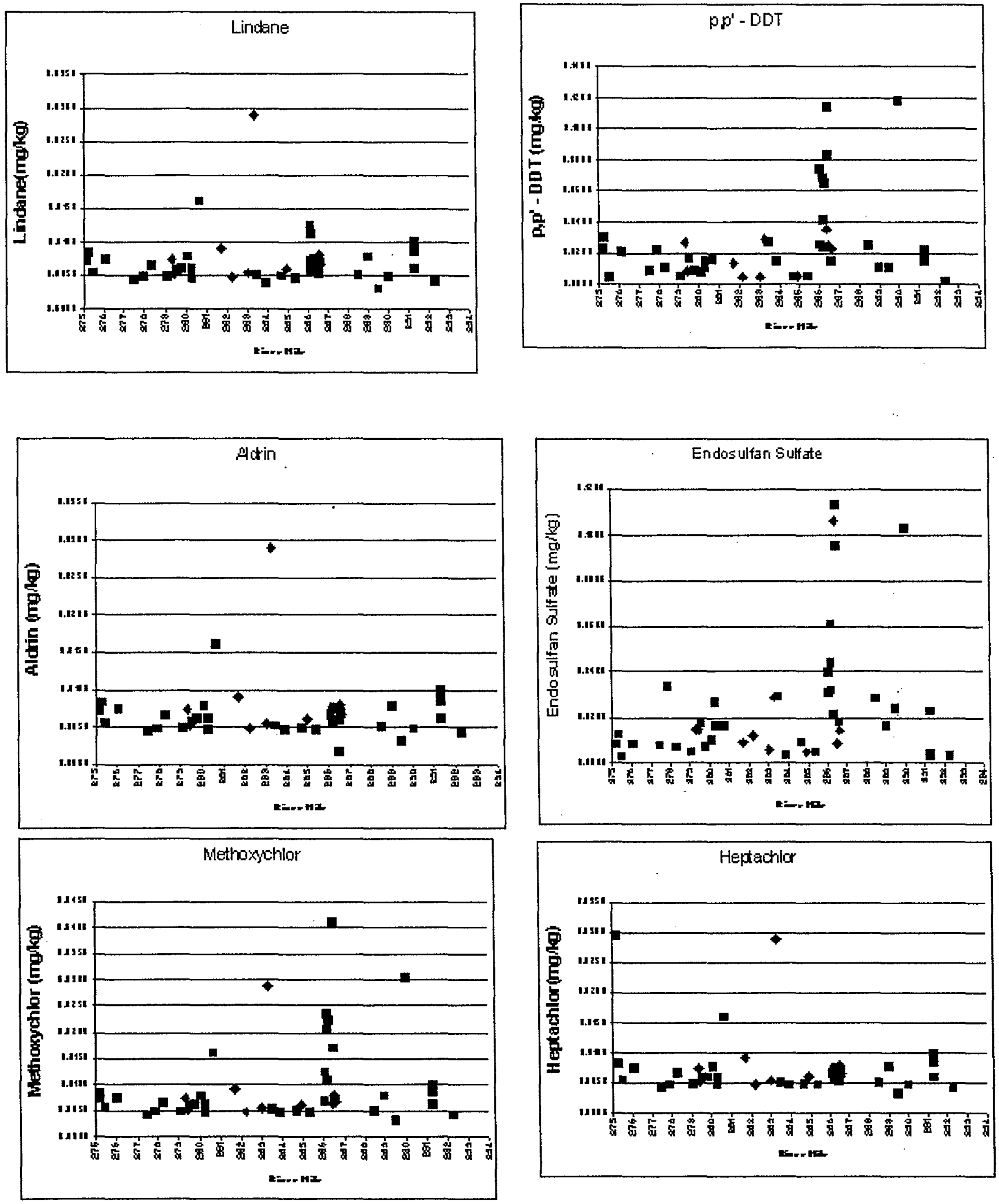
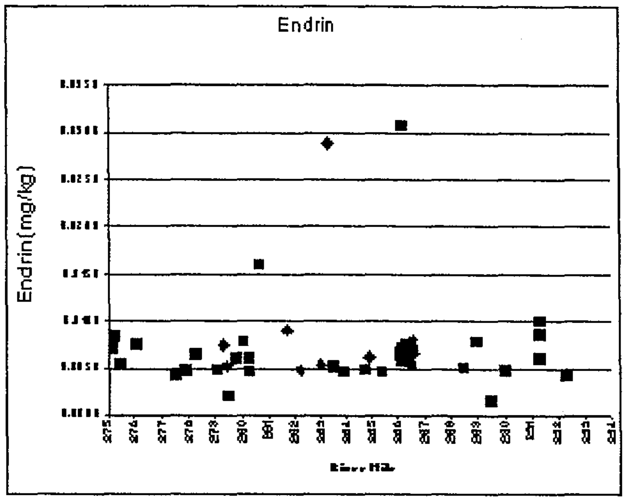
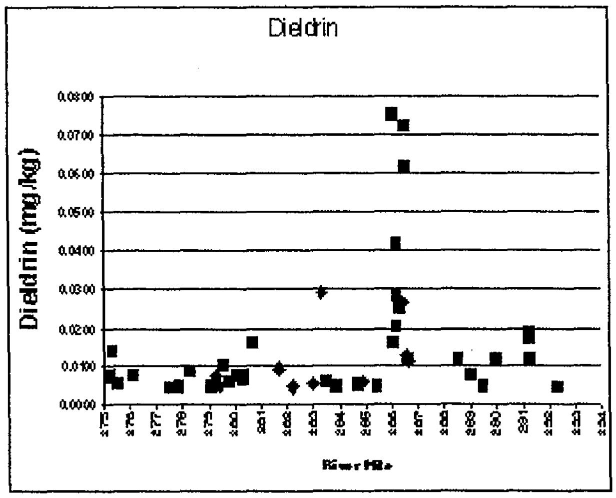
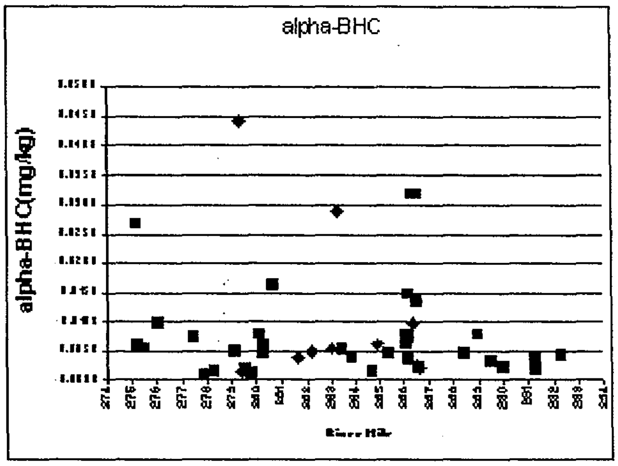
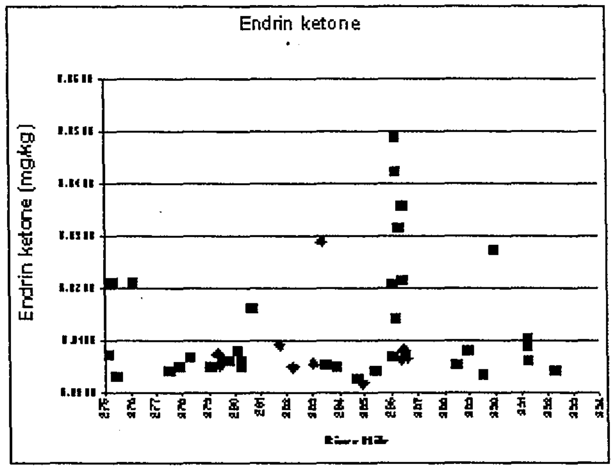


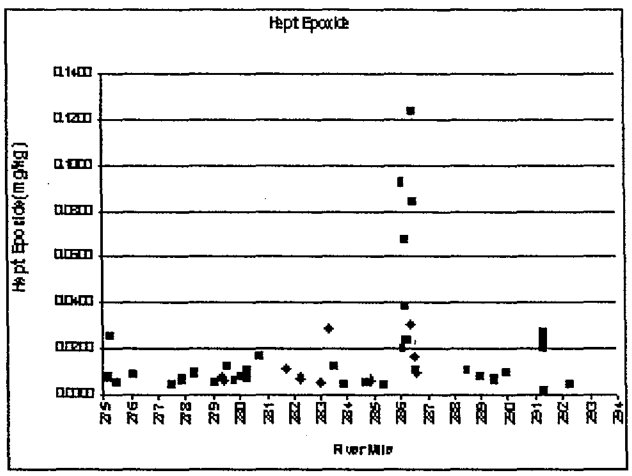
Figure 3.12 Concentrations of pesticides in the Lower des Plaines River Sediments - 2001 USEPA Survey











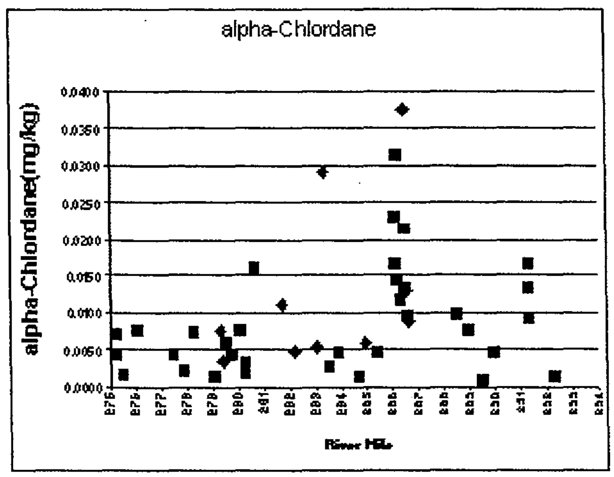


Figure 3.12 Continued

The analysis of pesticide contamination identifies three pesticides of concern: dieldrin (STU 10), heptachlor (STU 1.62) and heptachlor epoxide (STU 25.4). The STU of heptachlor of 1.6 signifies only mild contamination. It appears that most of the original heptachlor has been converted to heptachlor epoxide. The following discussion was taken from US EPA and FAO (Food and Agriculture Organization of the UN) web sites:

Dieldrin. The USEPA describes dieldrin as a byproduct of the pesticide Aldrin. From 1950 to 1974, aldrin and dieldrin were widely used to control insects on corn. Dieldrin was also used to control mosquitoes, as a wood preservative, and for termite control. Most uses of dieldrin were banned in 1987 and dieldrin is no longer produced in the US due to harmful effects on fish and wildlife. Dieldrin is persistent, bioaccumulative, and toxic. When released into the water system it does not undergo hydrolysis or biodegradation. It is subject to photolysis with a half-life of approximately four months, or somewhat faster in waters containing a photosensitizer.

Besides removing the contaminated sediment, either by currents or by dredging, or by capping it, no other feasible means of control are available. Based on the calculation of pore water, dieldrin in the sediment of the Lower Des Plaines River will not be acutely toxic to benthic organisms and will exhibit only mild chronic toxicity, mainly due to bioaccumulation.

Heptachlor and heptachlor epoxide. Heptachlor is an organochlorine cyclodiene insecticide, first isolated from technical chlordane in 1946. During the 1960s and 1970s, it was used primarily by farmers to kill termites, ants, and soil insects in seed grains and on crops, as well as by exterminators and home owners to kill termites. An important metabolite of heptachlor is heptachlor epoxide, which is an oxidation product formed from heptachlor by many plant and animal species.

Heptachlor is almost insoluble in water, and enters surface waters primarily though drift and surface runoff. In water and sediments, heptachlor readily undergoes hydrolysis to a compound that is then readily processed (preferentially under anaerobic conditions) by micro-organisms into heptachlor epoxide. After hydrolysis, volatilization, adsorption to sediments, and photodegradation may be significant routes for the disappearance of heptachlor from aquatic environments (Agency for Toxic Substances and Disease Registry, 1989). Heptachlor/heptachlor epoxide may be reduced with time; however, the half time is in years.

Other pesticides in the Lower Des Plaines River sediments were below the chronic toxicity levels. The very low chronic toxicity limits for the three pesticides of concern is primarily due to their bioaccumulation. The levels in the sediments are below the acute toxicity. However, they are likely impacting the composition and integrity of the benthic macroinvertebrate community (see Chapter 5).

Polychlorinated Biphenyls (PCBs)

Several PCB congeners were analyzed by the USEPA. Figure 3.13 shows the concentrations of PCBs in the Lower Des Plaines River sediments. Table 3.9 presents the calculations of the pore water concentrations. The USEPA water quality criteria do not list the acute (CMC) criterion.

Table 3.9 Calculation of Pore Water Concentrations of PCBs

PCB congener	CMC µg/L	CCC µg/L	Log K_{ow}	K_{ow} L/Kg	Π L/Kg	C_T µg/Kg	C_d µg/Kg	STU
1232-river (RM286)		0.014	4.5	34,673	1,092 (3,276)	600 6,000	0.55 1.83	**39** **130**
1221		0.014	4.09	12,302	387	600	1.54	**110**
1242-river (RM286)		0.014	4.11	12,882	405 (1,217)	1,000 16,000	2.46 13.14	**176** **939**
1254		0.014	6.03	1.07×10^6	33,752	1,000	0.029	**2.11**
1260		0.014	6.11	1.28×10^6	40,580	600	0.015	1.05

With the recent focus on remediation of sediment contaminated with PCBs, this preliminary analysis has now increased importance to the agencies involved (US Army Corps of Engineers, USEPA and IEPA).

The sediment concentrations of PCBs measured by the USEPA in the Lower Des Plaines River are high relative to some published benchmark values and the estimated pore water concentrations shown in Table 3.9. These concentrations may not be acutely toxic to benthic macro-invertebrates; however, PCBs bioaccumulate and biomagnify throughout the food chain. The highest concentrations of PCBs is the depositional zone above the Brandon Road Dam (RM 286+). The total PCB concentrations in the Lower Des Plaines River are similar to these measured in the Fox River (Wisconsin) downstream of DePere (WI). Because of its flow into Green Bay and ultimately into Lake Michigan, the Fox River has been studied for years and is now being remediated by the USEPA and Wisconsin Department of Natural Resources (1997). The sediment PCB concentration in the Sheboygan River in Wisconsin, put on the National Priority List, had in the 1980s in the impounded sections concentrations as high as 4,500 mg/Kg, which is two orders of magnitude greater than those measured at RM 286. Remediation of the Sheboygan River by excavation of the sediments brought the total PCB levels at post remediation to below 40 mg/Kg at the excavated sites (*Federal Register*, Vol. 51, No. 111, June 10, 1986), which is about the same as RM 286+ and ten times more than the PCB concentrations throughout the Des Plaines River UAA reaches. Other remediation projects, such as Waukegan Harbor, IL in Lake Michigan set the cleanup objectives levels for PCBs at 50 mg/Kg. Wisconsin DNR scientists also suspected that PCBs were transported from the PCB contaminated Cedarburg ponds to the Milwaukee River and Harbor attached to algal biomass. This could also be means of transporting PCBs from in the nutrient enriched reaches of the Lower Des Plaines and Illinois Rivers. It should be stated and emphasized that PCB concentrations throughout the most of the Lower Des Plaines River are below the existing objectives of clean up promulgated by the Illinois EPA and USEPA.

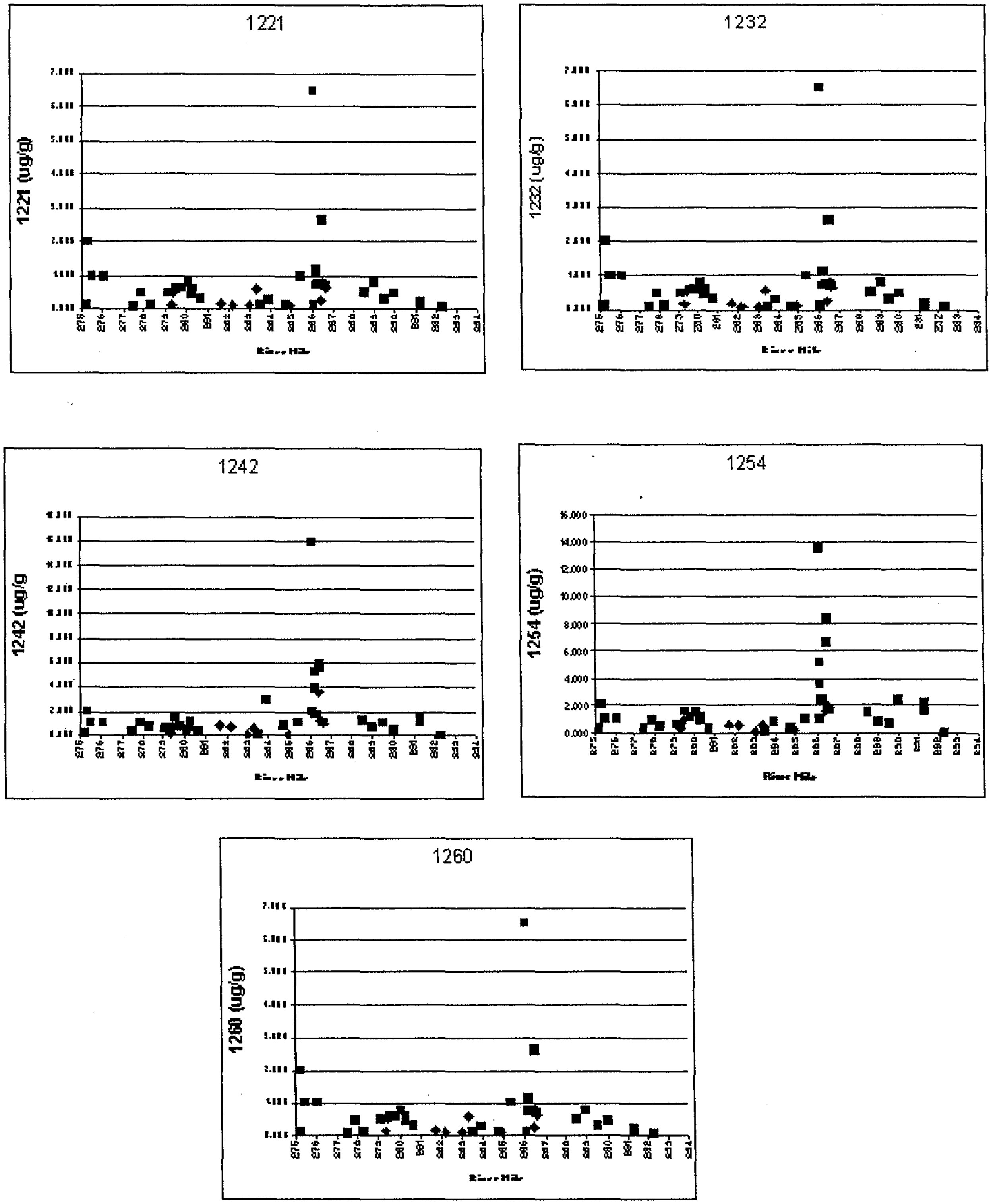


Figure 3.13 Concentrations of PCBs in the Lower Des Plaines River Sediments - 2001 US EPA surveys

PCBs are mixtures of different congeners of chlorobiphenyl, and the relative importance of the environmental fate mechanisms generally depends on the degree of chlorination. In general, the persistence of PCBs increases with an increase in the degree of chlorination. Mono-, di- and trichlorinated biphenyls biodegrade relatively rapidly, tetrachlorinated biphenyls biodegrade slowly, and higher chlorinated biphenyls are resistant to biodegradation. Although the biodegradation of higher chlorinated congeners may occur very slowly on an environmental basis, no other degradation mechanisms have been shown to be important in natural water and soil systems; therefore, biodegradation may be the ultimate degradation process. When released into water, adsorption onto sediment and suspended matter is an important fate process; PCB concentrations in sediment have been shown to be greater than in the associated water column. Although adsorption can immobilize PCBs (especially the higher chlorinated congeners) for relatively long periods of time, eventual dissolution into the water column has been shown to occur. The PCB composition in the water is enriched by the lower chlorinated PCBs because of their greater water solubility, and the least water soluble PCBs (highest chlorine content) remain adsorbed. In the absence of adsorption, PCBs volatilize from water relatively rapidly. However, strong PCB adsorption to sediment competes with volatilization, with the higher chlorinated PCBs having a longer half-life than the lower chlorinated PCBs. Although the resulting volatilization rate may be low, the total loss by volatilization over time may be significant because of the persistence and stability of the PCBs.

Polychlorinated biphenyls degrade into less-chlorinated PCBs that are more amenable to volatilization. PCBs have been shown to bioconcentrate significantly in aquatic organisms. Average log BCFs of 3.26 to 5.27, reported for various congeners in aquatic organisms, show increasing accumulation with the more highly chlorinated congeners. Making definitive conclusions on PCB bioaccumulation in the Lower Des Plaines River is difficult due to the absence of fish flesh analyses.

Other Priority Pollutants

Although the USEPA sediment analysis contains dozens of other organic and inorganic contaminants, only a few pollutants have a numeric standard/criterion for aquatic life protection. These are:

- Cyanides
- Pentachlorophenol
- Chlordane
- Gamma BHC, and
- Toxaphene

Cyanides in the sediment are mostly at or below the detection limit and are not a problem. Gamma BHC and toxaphene were not found in the USEPA sediment data base. Table 3.14 contains calculations of the pore water concentrations and STUs for the remaining two contaminants. Figure 3.14 presents the concentration plots pentachlorophenol, α-chlordane, oil and grease, ammonium and phosphate.

Nutrients are presented herein for documenting their levels. From the discussion on ammonium presented previously in this chapter, ammonium standards for water cannot be used to judge pollution content of sediments and there is no standard for phosphorus.

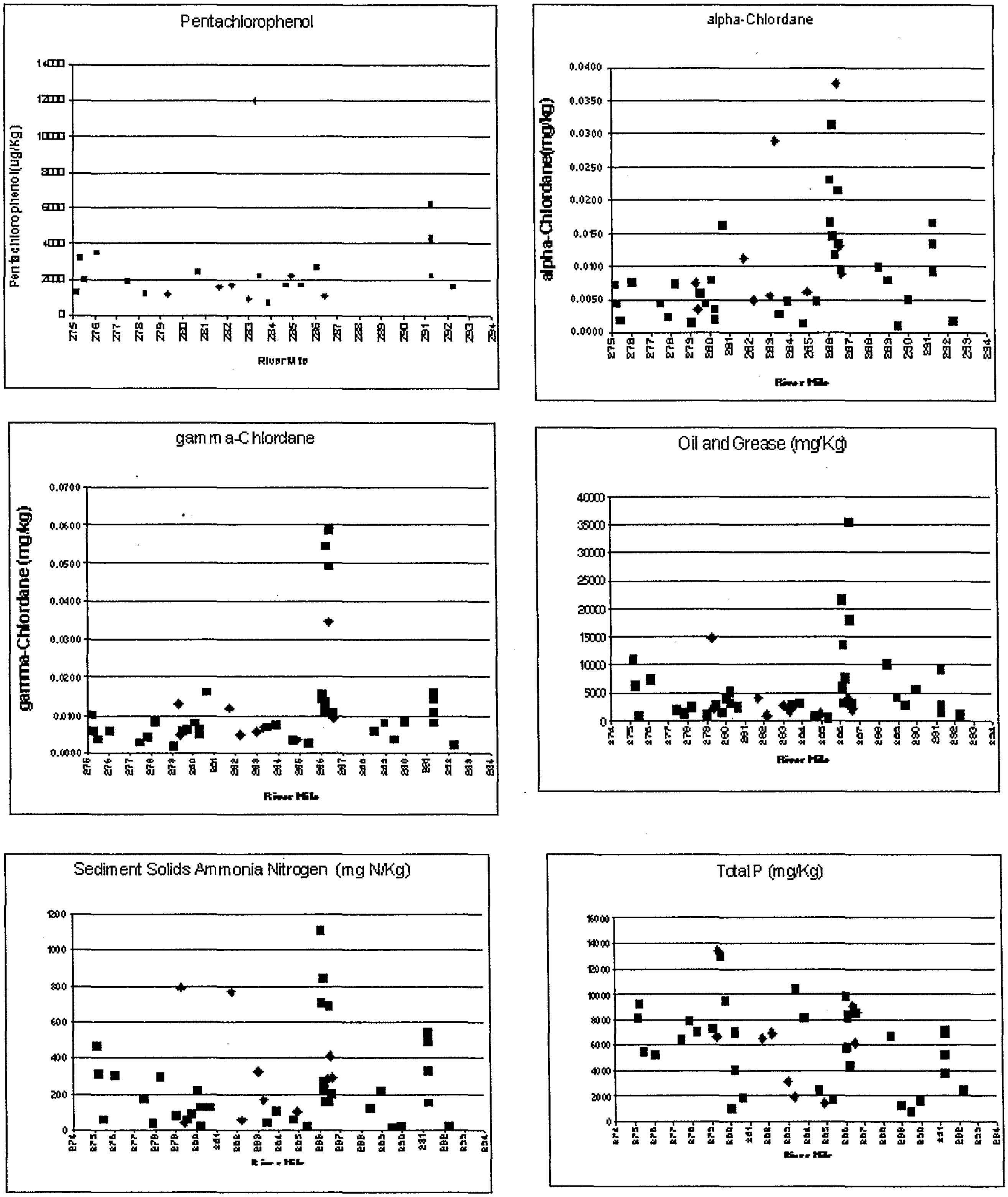


Figure 3.14 Concentrations of Pentachlorophenol, Chlordane, Oil and Grease, Ammonium and Phosphate

Table 3.14 Calculation of pore water concentrations of pentachlorphenol and chlordane

Contaminant	CMC µg/L	CCC µg/L	Log K_{ow}	K_{ow} L/Kg	Π L/Kg	C_T µg/Kg	C_d µg/L	STU
Pentachlo-phenol	20	13	5.01	102,329	3,223	2,000	0.62	0.05
Chlordane	24	0.0023	2.78	602	18.98	5	0.26	**114**

Chlordane. The calculation suggest chlordane may pose a problem in the Lower Des Plaines River sediments, in spite of its very low concentration. Chlordane is highly persistent, does not chemically degrade and is not subject to biodegradation in soils. Chlordane molecules usually remain adsorbed to clay particles or to soil organic matter. Chlordane does not degrade rapidly in water. It can exit aquatic systems by adsorbing to sediments or by volatilization.

The photoisomers of chlordane are of special significance because to certain animals they are much more toxic than chlordane. Photo-cis-chlordane, which is more biodegradable than cis-chlordane, showed higher bioaccumulation values and therefore may have more significant effects on food chains.

Evaporation is the major route of removal from soils and aquatic systems. The volatilization half- life of chlordane in lakes and ponds is estimated to be less than ten days. However, adsorption to sediment significantly attenuates the importance of volatilization. Chlordane is thought to have a high bio-accumulation in aquatic organisms.

Polycyclic Aromatic Hydrocarbons (PAHs)

Much interest in the environmental community has been devoted to this class of contaminants. Unfortunately, no numeric water or sediment criteria have been issued for aquatic life and the issue of toxicity in streams cannot be adequately addressed. The toxicity of PAHs to aquatic organisms is generally low, i.e., known LC(50) values are relatively high.

Polycyclic aromatic hydrocarbons (also known as polynuclear aromatic hydrocarbons) are composed of two or more aromatic (benzene) rings which are fused together when a pair of carbon atoms is shared between them. The environmentally significant PAHs are those molecules which contain two (e.g., naphthalene) to seven benzene rings. In this range, there is a large number of PAHs which differ in the number of aromatic rings, position at which aromatic rings are fused to one another, and number, chemistry, and position of substituents on the basic ring system.

Physical and chemical characteristics of PAHs vary with molecular weight. For instance, PAH resistance to oxidation, reduction, and vaporization increases with increasing molecular weight, whereas the aqueous solubility of these compounds decreases. As a result, PAHs differ in their behavior, distribution in the environment, and their effects on biological systems. PAHs can be divided into two groups based on their physical, chemical, and biological characteristics. The lower molecular weight PAHs (e.g., 2 to 3 ring group of PAHs such as naphthalenes, fluorenes, phenanthrenes, and anthracenes) have significant acute toxicity to aquatic organisms, whereas the high molecular weight PAHs, 4 to 7 ring (from chrysenes to coronenes) do not. However, several members of the high molecular weight PAHs have been known to be carcinogenic.

Among a large number of compounds in the category of polycyclic aromatic hydrocarbons, only a few are manufactured in North America. These PAHs are mostly used as intermediaries in pharmaceutical, photographic, and chemical industries. Naphthalenes are also used in the production of fungicides, insecticides, moth repellent, and surfactants. PAHs are also formed during pyrolysis in coal gasification plants, which used to be common in the Chicago area and in coke plants, which continued to operate in the area until a few years ago. PAHs are also emitted in exhausts of Diesel engines. Significant concentrations of PAHs can be found in soils near highways and in streams receiving urban and highway runoff (Novotny et al., 1999; Novotny, 2003). A comprehensive compilation on PAHs toxicity in aquatic systems has been published by the Ministry of the Environment, Land and Parks of British Columbia (Nagpal, 1993) downloadable from Internet. Most of the discussion and information on acute and chronic toxicity is taken from this document.

As it is with other potentially toxic compounds, PAHs toxicity is related to: (a) the PAH type, (b) the species exposed, (c) the duration and the type of exposure. The higher molecular weight PAHs (containing more than 3 aromatic rings) such as benzo[a]anthracene and benzo[a]pyrene, have shown to be acutely toxic to benthic invertebrates at relatively low concentrations (5-10 μg/L). However, such dissolved concentrations in natural systems may not be achievable because of the very large magnitude of the partition coefficient for PAHs. Alkyl homologues of PAHs are generally more toxic to aquatic life than the parent compound. For instance, the 48-h EC for *Daphnia pulex* exposed to anthracene (750 μg/L) was much higher than that obtained when the organisms were exposed to methyl anthracene (EC(50) = 96 μg/L) or methoxy anthracene (EC(50) = 400 μg/L).

Table 3.15 assesses the PAH pollution. PAH concentrations are then plotted on Figure 3.15. No statutory aquatic life standards are available and the toxic limits were obtained from Nagpal (1993). The smallest LC(50) for the most sensitive species was included in the table. Most LC(50) values in Table 3.15 were 48 to 96 hrs. Based on the ratio of pore water concentration/LC(50), a judgement was made on the magnitude of the STU. Table 3.15 documents that PAHs in the Lower Des Plaines River sediments would not be toxic to such sensitive organisms as rainbow trout.

Figures 3.15 also shows that at River Mile 286+ (upstream of the Brandon Road Dam) the concentrations of some PAHs were about five to ten times greater than throughout the rest of the reaches. However, this will not change the conclusion on the toxicity because the sediment at this location has higher organic carbon content (about three times more), resulting in about 3 times larger partition coefficient.

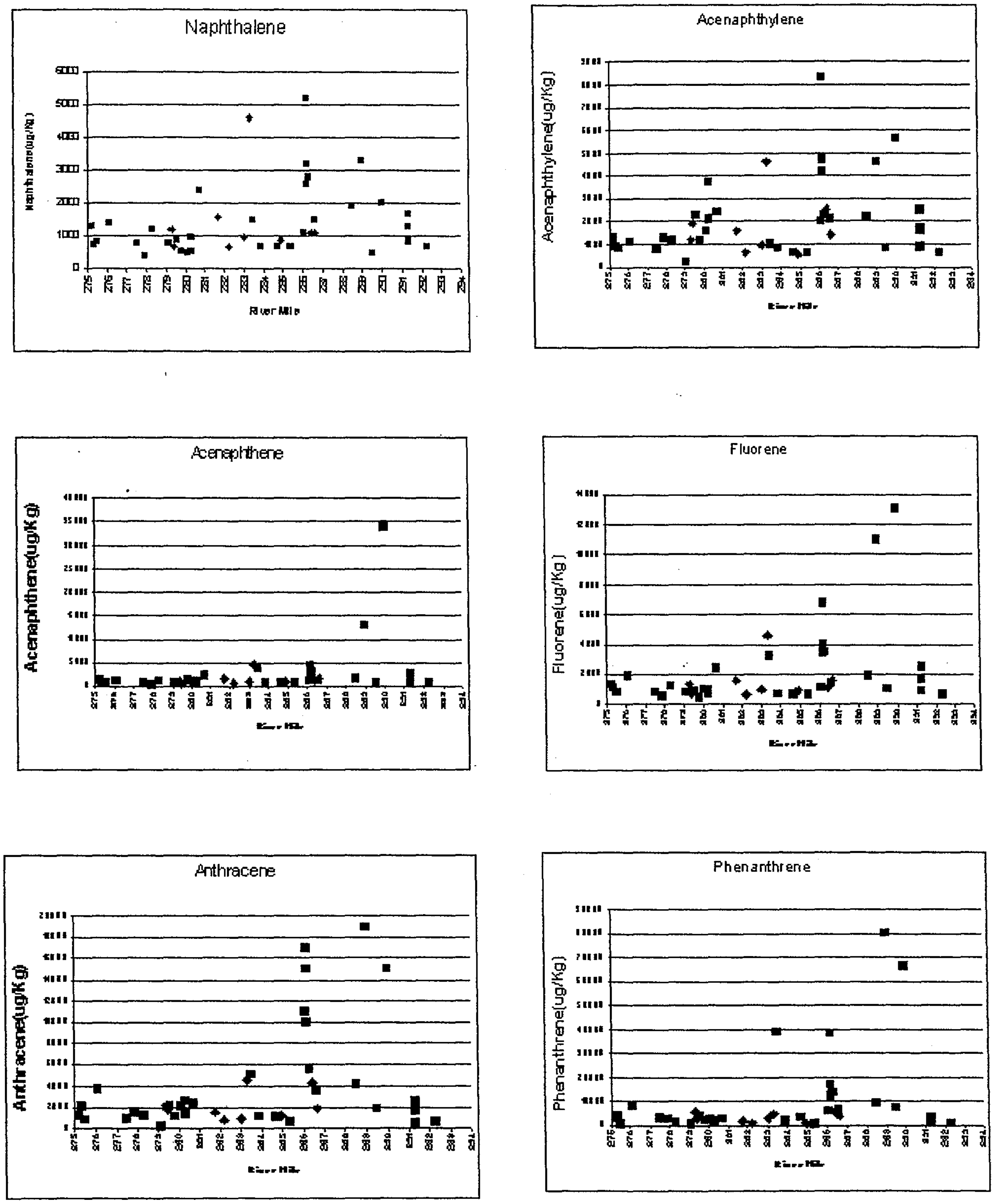
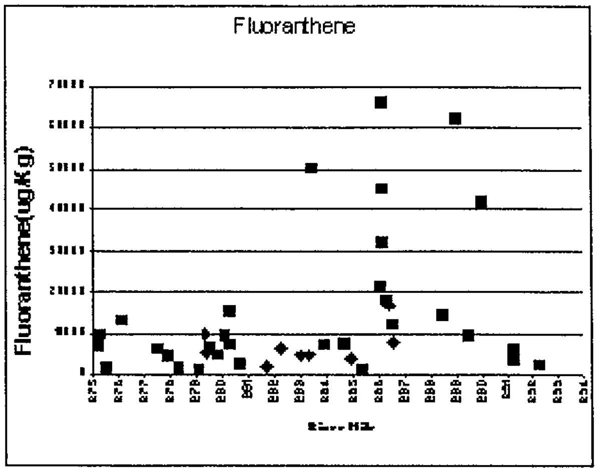
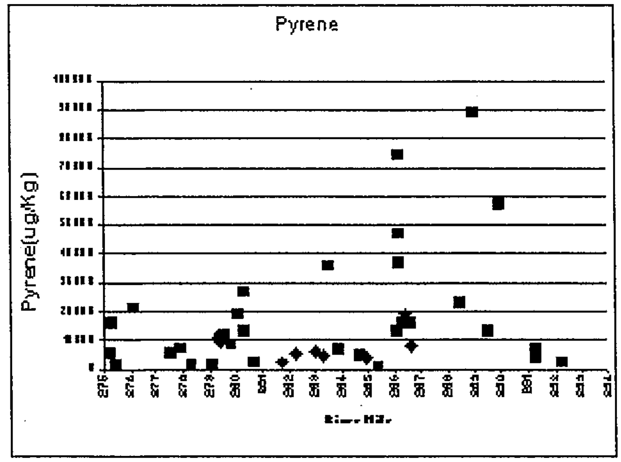
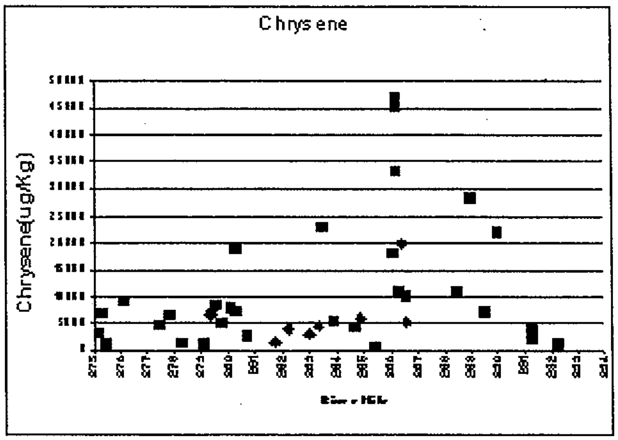
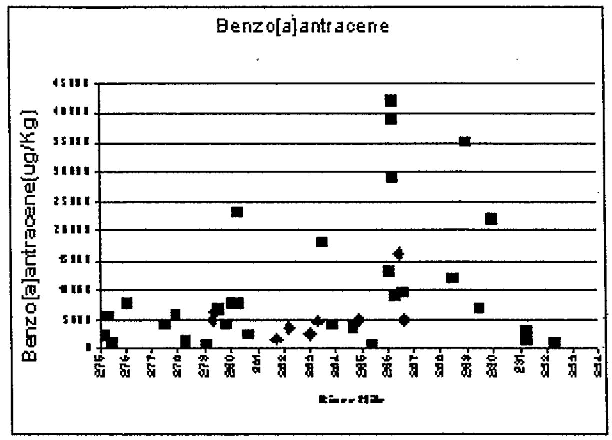


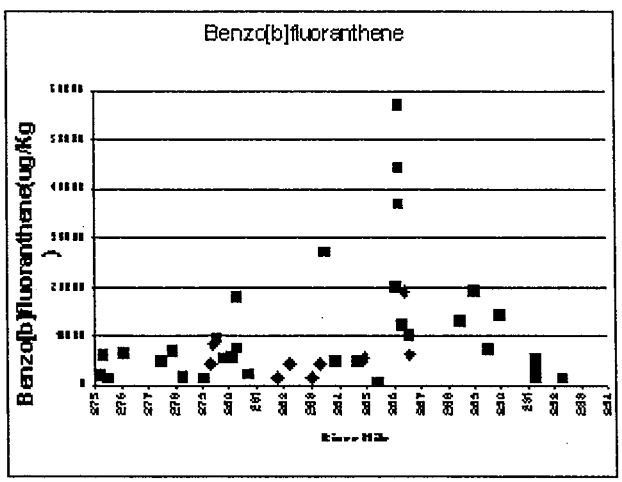
Figure 3.15 PAHs Concentrations in the Lower Des Plaines River











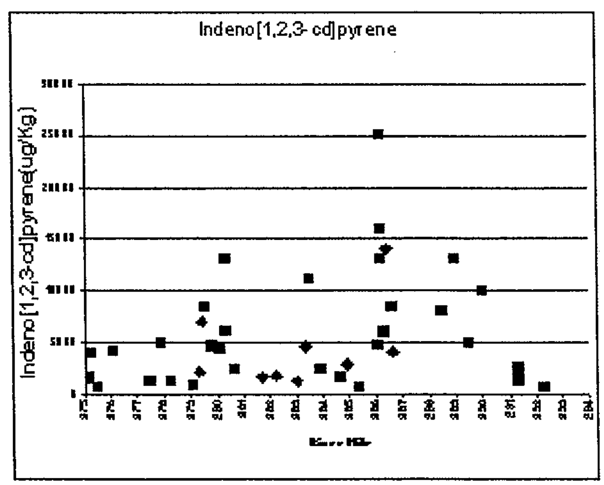


Figure 3.15 Continued

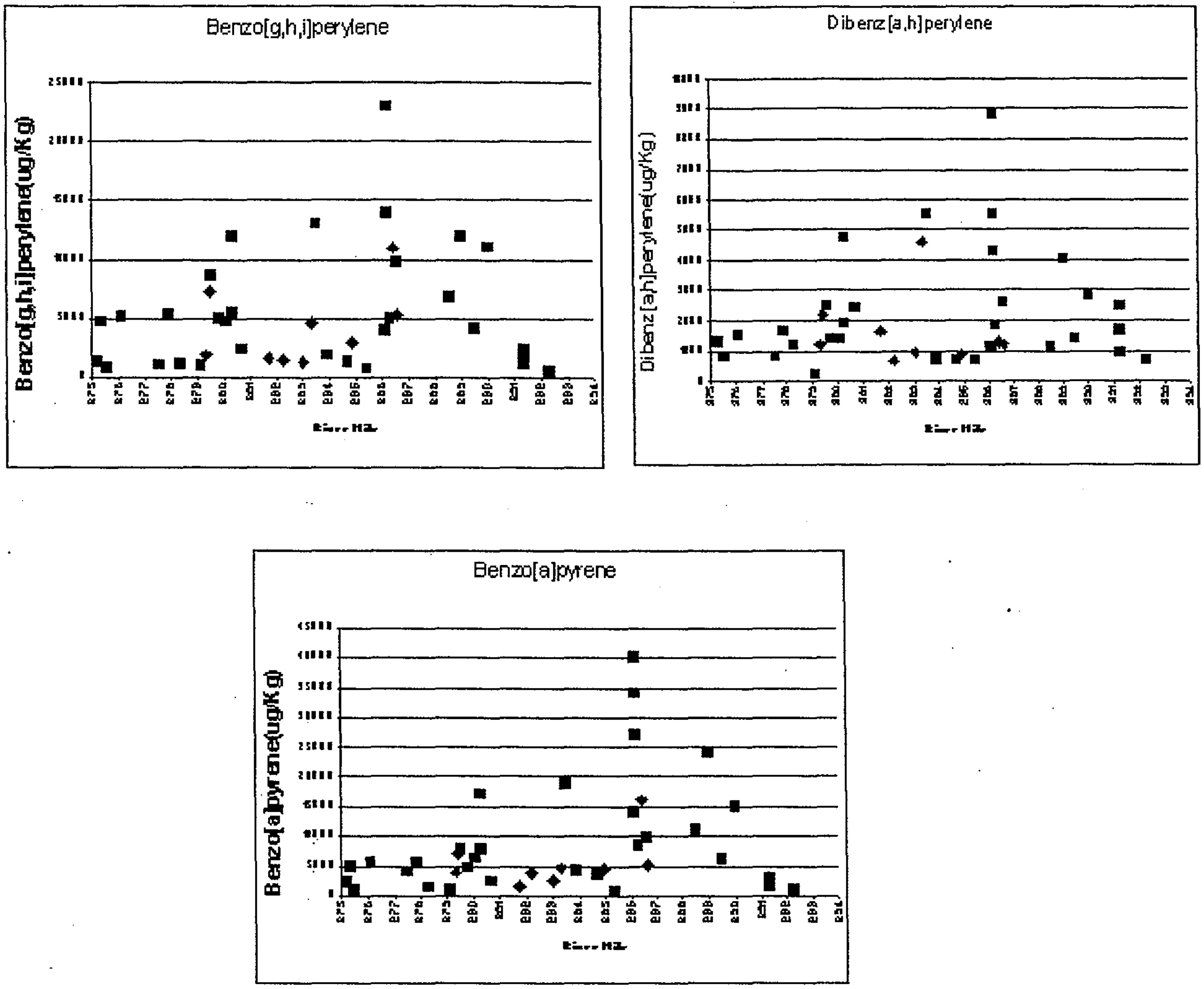


Figure 3.15 Continued

The British Columbia Ministry of Environment, Lands and Parks report by Nagpal (1993) also states that upon exposure to sunlight (ultraviolet light) the PAHs accumulated by aquatic organisms have been shown to be severely toxic. This confirms Burton's (1995) experiments with the sediment from the Brandon Road Dam tailwaters. Nagpal quoted experiment by Bowling et al., (1983) who found that 12.7 μg/L of anthracene was fatal to bluegill sunfish in 48 hours in an outdoor channel in bright sunlight. No mortality was noted when the test organisms (bluegill sunfish) were in the shaded area of the channel. It was concluded that the direct sunlight exposure of anthracene-contaminated organisms and not the toxic anthracene photoproducts in water, was responsible for the mortality of the bluegill.

The photo-induced effects may have little relevance in the Lower Des Plaines River because of the depth of the sediments and turbidity of water. Furthermore, concentrations were calculated for pore water of sediments and its effect on benthic macroinvertebrates and do not reflect the concentrations of these compounds in water and their effect on fish and other water living organisms. Nevertheless, photo-activation in very shallow portions of the river may have some adverse effects on the benthic organisms. However, overall, PAHs may not be contaminants of concern in the Lower Des Plaines River.

Table 3.15 Calculation of pore water concentrations of PAHs

PAH	LC(5) μg/L	Chronic limit μg/L	Log K_{ow}	K_{ow} L/Kg	Π L/Kg	C_T μg/Kg	C_d μg/L	STU
Acenaphthene	600	-	3.92	8,318	262	2,000	7.6	<<1
Anthracene	360		4.45	28,184	888	2,000	2.25	<<1
Fluoranthene	200		5.33	213,796	6,734	10,000	1.48	<<1
Fluorene	210	125	4.18	15,136	477	2,000	4.2	<<1
Benzo(a)anthracene	10		5.61	407,380	12,832	5,000	0.38	<<1
Napthalene	120+		3.37	2,344	74	900	12.2	<1
Phenathrene	30+		4.46	28,840	908	4,000	4.4	<1
Benzo (a)pyrene	5		5.98	0.95×10^6	30,082	5,000	0.16	<<1

+ Long term exposure (648 hrs) of rainbow trout.

Conclusions on Sediment Contamination

We have presented an extensive analysis of trends and effects of contaminants present in the past and currently in the sediments of the Lower Des Plaines River. We have found that the quality of sediments has been improving over the years; however, we have also specified the following concerns, mainly based on the most recent USEPA detailed analysis of the sediments in the investigated reaches:

- ☐ The USEPA survey and our analysis identified an area of contaminated sediments in the depositional zone above the Brandon Road Dam (RM 286+). The sediment has high PCB, pesticide and elevated toxic metal contamination relative to benchmark and background levels. PCB pore water contamination exceeds by two orders of magnitude the PCB chronic criteria for water. We did not and could not identify sources of PCBs; it is most likely a legacy pollution originating from multiple sources years ago. The PCB concentrations throughout the most of the Lower Des Plaines River are below the existing objectives of clean up promulgated by the Illinois EPA and USEPA.
- ☐ The Lower Des Plaines River sediments also have high concentrations of dieldrin, chlordane and heptachlor epoxide. Dieldrin and heptachlor epoxide are toxic byproducts of biological degradation of pesticides used years ago. All three pollutants are very high in the RM 286+

depositional zone. Again this pollution is characterized as legacy pollution. These pesticide pollutants were used years ago as insecticides on crops, in homes and other wide spread uses.

- ☐ Higher temperatures in the Upper Dresden Island pool may also have some effect on the quality of the sediment as it impacts toxicity of ammonium and may directly affect benthic organisms. On the other hand, temperature may enhance pollutant degradation in the sediments. However, we have not found ammonium in the sediment to be a source of toxicity to organisms residing in the interstitial benthic layer or bottom feeders.

- ☐ Toxic metals do not appear to be a toxicity problem with the exception of cadmium in the RM 286+ depositional zone.

- ☐ Individual PAHs are generally not a toxicity problem.

- ☐ The USEPA measured dozens of other pollutants but for the lack of a criterion or a guidance we could not perform an adequate assessment. Most of these pollutants (e.g., aromatic and chlorinated hydrocarbons) have relatively high LC(50) for aquatic organisms and may not be a problem at measured levels. However, the USEPA 2001 sediment contamination database is very large and necessitates further detailed analysis in order to completely identify other possible organic contaminants and synergetic effects.

- ☐ The toxicity of sediments due to PCBs and two byproducts of pesticide degradation and symbiotic effects of all remaining contaminants will reflect on the composition and integrity of benthic and bottom feeding organisms due to mostly chronic toxicity effects. Potential benthic macroinvertebrate and effects will be documented in Chapters 5 and 6 of this report.

- ☐ Elevated PCB concentrations, due to the biomagnification in the food chain, may also be reflected in fish and water fowl tissue contamination. However, no measurements were provided to us by the agencies. Such analyses should be a part of the proposed sediment remediation study.

- ☐ A more definitive evaluation of sediment toxicity is not possible without sediment bioassay and fish analysis data, which are currently limited or lacking. Reports of the widespread presence of sludge worms sensitive to some potentially toxic compounds suggest that the toxics may be tied up in the sediment complex and not necessarily available to the biota.

<u>UAA Issues</u>

(1) Naturally occurring pollutant concentrations prevent attainment of the use;

The pollutants of concern, PCBs and pesticide byproducts, are strictly human made products and pollutants. Reason # 1 does not apply.

(2) Natural, ephemeral, intermittent or low flow or water levels prevent the attainment of the use unless these conditions may be compensated for by the discharge of a sufficient volume of effluent discharge without violating state conservation requirements to enable uses to be met;

Reason # 2 is not applicable.

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place;

Resolution of the sediment contamination problem will require a study determining the methods of remediation (e.g., dredging and disposal, capping, or allowing time to take care of the problem). With today's state of the art of contaminated sediment remediation, solving the problem is feasible.

(4) Dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use;

There is no doubt that the high contamination of sediments in the depositional zones, especially in the RM 286+ depositional zone, is caused by impounding the river for navigation. Impounding for navigation is a physical condition that is irreversible in the long run. On the other hand, continuous scouring and resuspension by barge traffic may have a cleansing effect.

(5) Physical conditions related to the natural features of the water body, such as the lack of proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses;

This issue pertaining to habitat and bottom sediments will be addressed in Chapters 4 and 6.

(6) Controls more stringent than those required by Sections 301(b)(1)(A) and (B) and 306 of the Act would result in substantial and wide-spread adverse social and economic impact.

The pollutants of concerns are most likely legacy pollutants and more stringent controls of current effluents will not remedy the problem. The production and use of these pollutants were outright banned more than twenty years ago. The required actions are in the category of stream restoration and remediation that may rely, if the responsible sources of the contamination are not identified, on public financing. In this case consideration of a wide spread adverse socio - economic impact may be needed; however, recent cases of completed or planned remediation of many sites contaminated by PCBs, including Hudson River in the State of new York, the Sheboygan and Fox River in Wisconsin, Cedar Creek in Wisconsin may provide a precedent indicating that a wide spread socio-economic impact may not occur.

We are proposing to the responsible agencies (IEPA, USEPA, US Army Corps of Engineers) to conduct an interagency study on the extent of sediment contamination of the Lower Des Plaines River that would build upon the USEPA survey and monitoring by IEPA and

MWRDGC and the Midwest Generation sediment study by Burton. The proposed study should identify the sources of these contaminants, the rate of recovery, the extent of contamination within the reach and upstream and, above all, sources of financing of the remediation plan. The study should be conducted at the conclusion of the Use Attainability Analysis of the Chicago Area Waterway System.

This study of the sediments should not delay implementation of attainable standards for the water column.

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CHAPTER 4

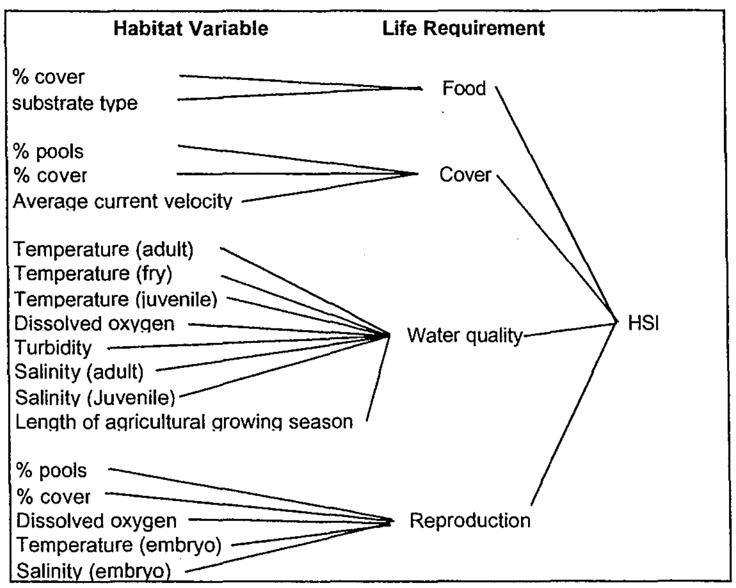
PHYSICAL HABITAT OF THE LOWER DES PLAINES RIVER

Introduction

In Section 101(a) of the Federal Clean Water Act (CWA), it is stated that its "*the Congressional declaration of goals and policy*" to achieve "*Restoration and maintenance of chemical, physical and biological integrity of Nation's waters...*". A growing body of literature has documented that factors other than chemical water quality may be responsible for the resultant conditions of the stream ecosystem (Karr and Dudley, 1981; Karr et al., 1986; Rankin and Yoder 1990; Rankin 1995, Yoder and Rankin, 1995). A stream is a complex ecosystem in which several biological, physical and chemical processes interact. An important factor determining the presence and abundance of aquatic organisms is physical habitat (Gorman and Karr, 1978; Schlosser, 1982).

Habitat can be defined as the total chemical and physical environment where organisms live. Figure 4.1 summarizes the relationship between habitat and biological condition as measured with a Habitat Suitability Index (HSI).

FIGURE 4.1
The Relationship between Habitat and Biological Condition
Source USEPA, 1989



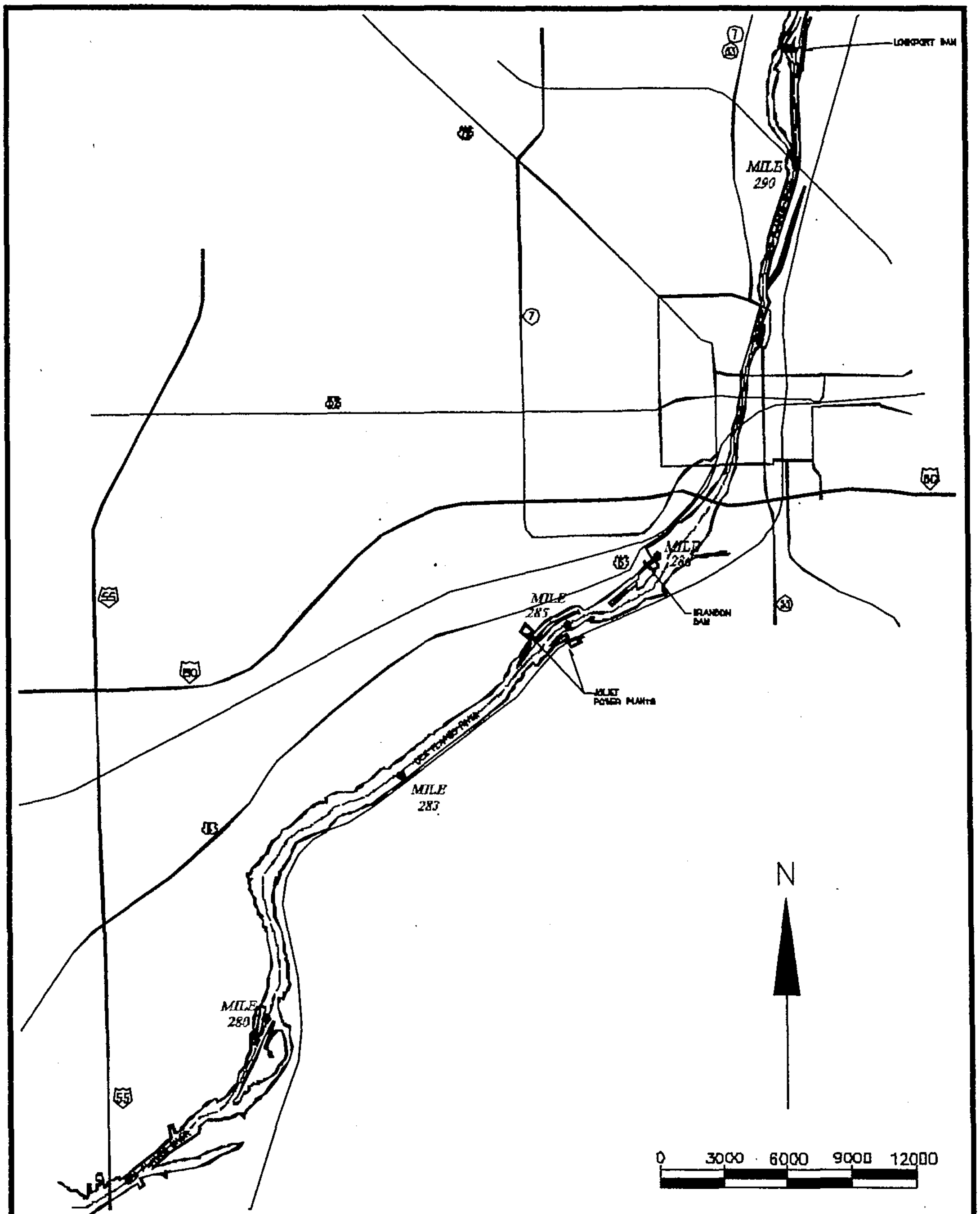
Several authors have proposed methodologies for assessing habitat in streams, including such methodologies as the Habitat Suitability Index (Terrell, 1984), Habitat Quality Index (Binns and Eiserman, 1979), Physical Habitat Simulation Model (PHABSIM) (Hilgart, 1982), Rapid Bioassessment Protocols (RBP) (USEPA, 1989), and the Qualitative Habitat Evaluation Index (QHEI) (Rankin, 1989). The firm, EA Engineering, Science and Technology, on contract with Commonwealth Edison Company, a discharger to the Lower Des Plaines River, conducted a habitat assessment of the Use Attainability Analysis (UAA) study area in 1993 and 1994 (Commonwealth Edison, 1996). The study used the Qualitative Habitat Evaluation Index (QHEI) developed by the State of Ohio EPA (Rankin, 1989). This available data will be used here to help define the physical habitat of the Lower Des Plaines River.

The following chapter will describe the current physical aquatic habitat of the Lower Des Plaines River and its relationship to maintaining habitat for fish and aquatic life. The physical characteristics will be described and the results of the QHEI inventory will be summarized.

Study Reach

The study area for the Use Attainability Analysis of the Lower Des Plaines River extends from the confluence of the Des Plaines River with the Chicago Ship and Sanitary Canal (CSSC) at the E.J.& E. railroad bridge (River Mile 290.1 near Lockport) downstream to the I-55 Highway Bridge at River Mile 277.9 (Figure 4.2). Almost the entire reach is impounded and has two morphologically different segments, the Brandon Road Pool above the Brandon Road Lock and Dam (River Mile 286) and the portion of the Dresden Island Pool above the I-55 Bridge. The Brandon Road Lock and Dam physically separate the two segments. The dams are operated by the U. S. Army Corps of Engineers.

The Brandon Road Pool is four miles in length, approximately 300 feet wide, with the depth varying between 12 - 15 feet. The Dresden Island Pool is 14 miles long, approximately 800 feet wide, with the depth varying between 2 - 30 feet. Table 4.1 summarizes some of the geographical features in the study reach by river mile.



Hey and Associates, Inc. Water Resources, Wetlands and Ecology	*Des Plaines River*	Figure 4.2 UAA Study

TABLE 4.1
Geographical Features in the Lower Des Plaines River Study Area

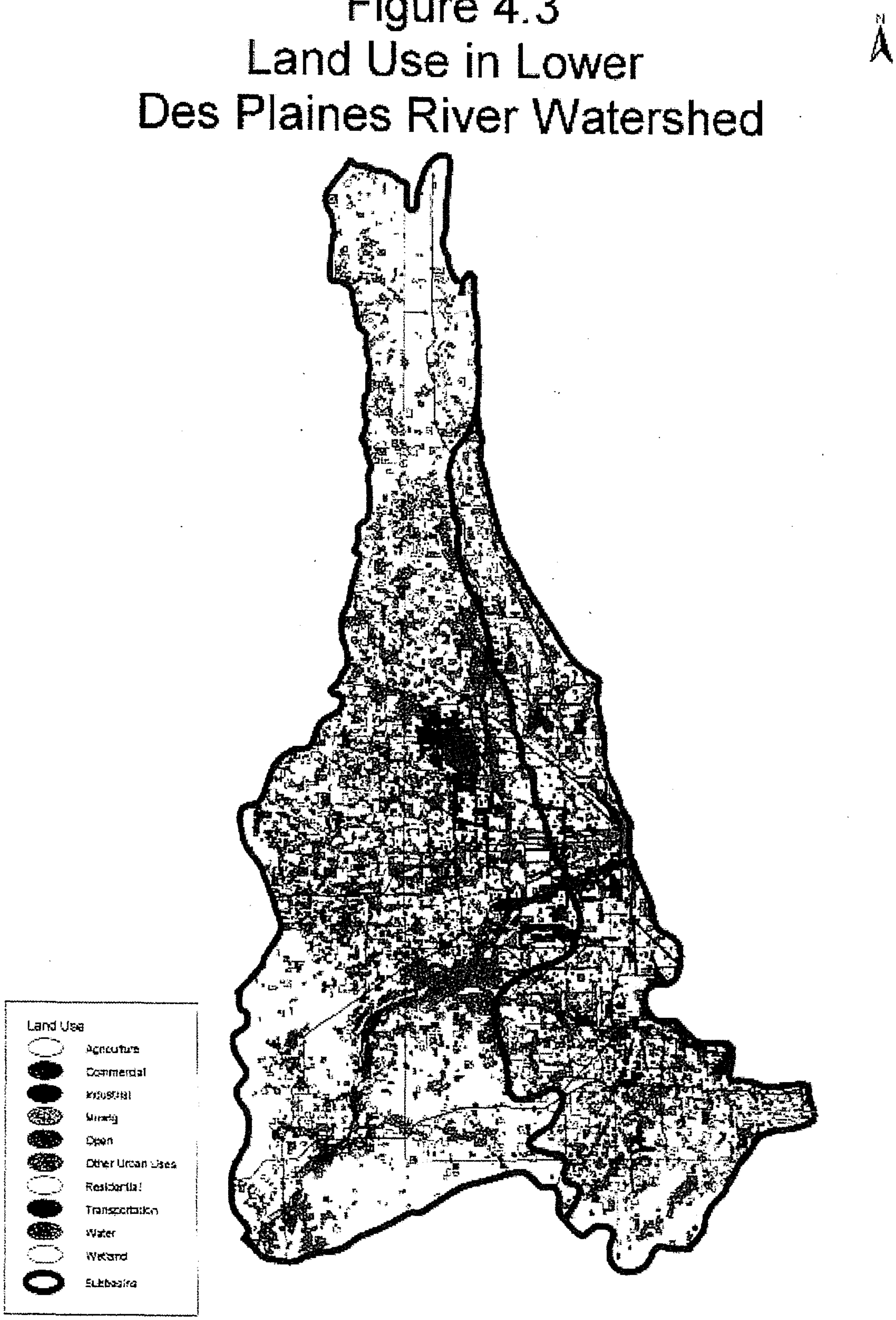
Feature	River Mile
Dresden Island Lock and Dam	271.5
I-55 Bridge	277.9
Mouth Jackson Creek	279.2
Treats Island	279.5
Mouth Cedar Creek	280.0
Midwest Generation Joliet Power Plants	284.3 & 284.7
Brandon Road Loc and Dam	286.0
I-80 Bridge	286.9
Jefferson Street – Joliet	287.9
Harrah's Casino	288.0
Ruby Street – Joliet	288.3
E.J.& E Railroad Bridge	290.1
Confluence Des Plaines River with Chicago Sanitary and Ship Canal	290.1
Lockport Lock and Dam	290.8

Watershed Characteristics

The drainage area of the study reach is approximately 1,500 square miles at Joliet, Illinois. Of this drainage area, 843 square miles is made up of the Upper Des Plaines River and 657 square miles is from the Chicago Sanitary and Ship Canal (CSSC). Based on USGS records from gauging stations at Lamont, on the Des Planes River, and Romeoville, on the CSSC, the annual mean flow at Joliet is approximately 4,450 cfs. The make up of this annual mean flow is approximately 3,510 cfs from the CSSC, and 940 cfs from the Upper Des Plaines River. Of the entire annual mean flow in the Lower Des Plaines River approximately 1,880 cfs (42.%) is made up treated wastewater effluent. Base flow is approximately 2,700 cfs and 350 cfs from the CSSC and Upper Des Plaines River respectively. The average stream velocity at mean flow is 0.65 fps.

Land use for the Des Plaines River Watershed, which includes the Upper and Lower Des Plaines River and the Chicago Sanitary and Ship Canal, is summarized in Table 4.2 and illustrated in Figure 4.3. The study area watershed is dominated by urban development, which makes up 50.5 percent of the watershed. Agriculture is the second most dominant land use and is located predominantly in the headwaters area of the Upper Des Plaines River in Wisconsin.

Figure 4.3
Land Use in Lower Des Plaines River Watershed



Not to Scale

TABLE 4.2
Land Use in the Des Plaines River Watershed

Land Use	Area (Square Miles)	Percent
Commercial	188.8	9.1
Industrial	88.6	4.3
Residential	575.0	27.6
Transportation	107.3	5.2
Other Urban Uses	91.9	4.4
Mining	12.9	0.6
Agriculture	838.8	40.3
Open Space	141.0	6.8
Wetlands	17.2	0.8
Water	20.5	1.0
Total	**2082**	**100.0**

Source: USEPA, 1999

The study area is located in the Central Corn Belt Plains ecoregion. Ecoregions are areas of relatively homogenous ecological systems or relationships between organisms and their environment (Omernik, 1987). Ecoregion classification builds on single-purpose geographical classifications (such as physiography, climate, or soils) to create a framework for understanding regional patterns. The homogenous ecological system concept becomes important when selecting reference sites for comparison of biological data as will be discussed in Chapters 6 and 7 of this report.

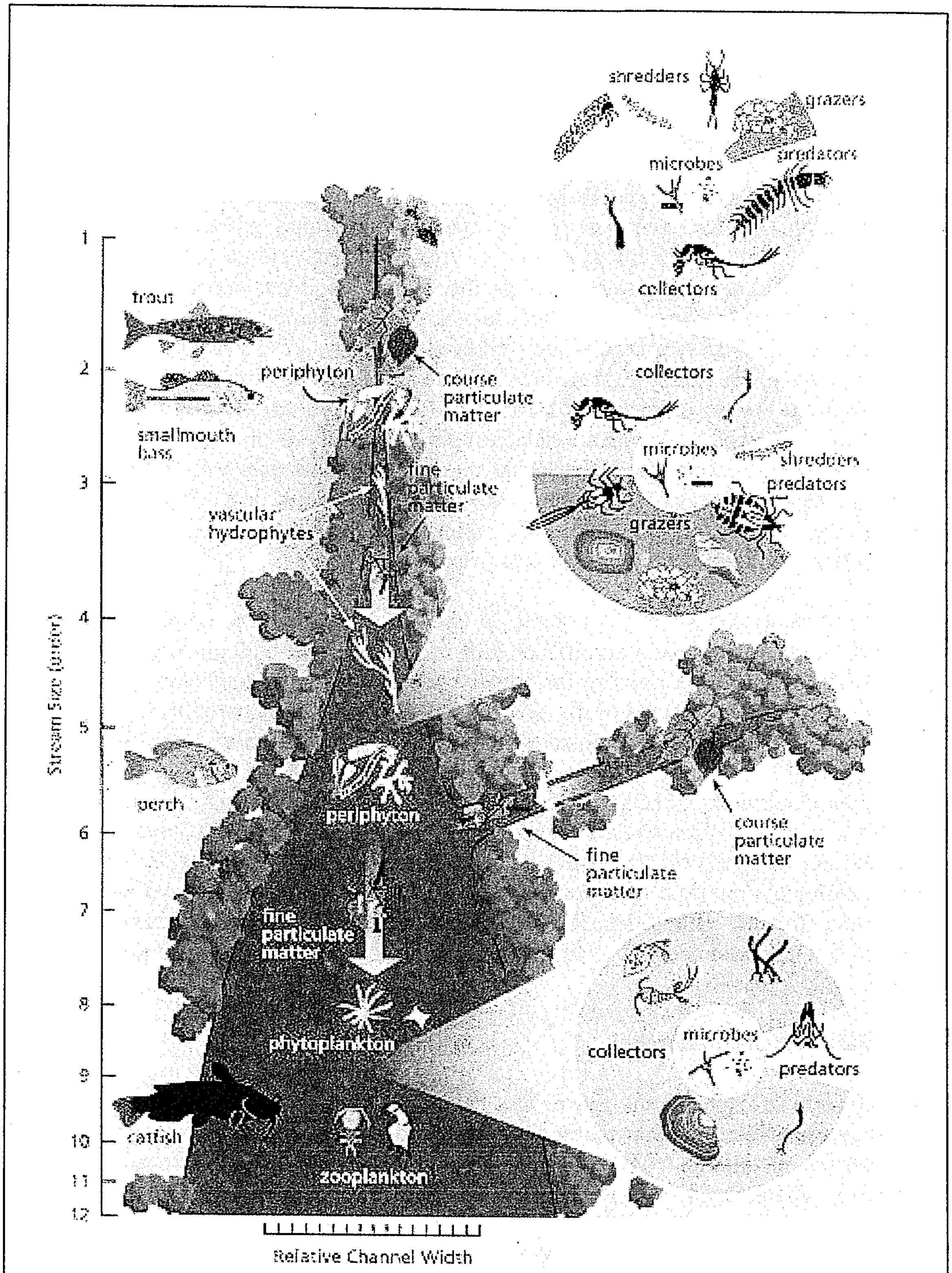
Historically, vegetation in the Central Corn Belt Plains was a mosaic of bluestem prairie and oak-hickory forest (USGS, 1999). The prairie covered the flat uplands and the forest typically occupied stream valleys and moraines. High stream turbidity and sedimentation are documented problems in the streams of the Central Corn Belt Plains ecoregion (Omernik and Gallant, 1988).

The upper most bedrocks in the study area are dolomite and limestone, which is the cause of the high alkalinity and hardness of the local waterways. Surficial geologic materials are made up of sand, and sand gravel deposits left by five major glacial periods. Typical glacial features such as till and outwash plains, moraine, kettles, kames and drumlins are found in various part of the Lower Des Plaines River watershed. Bed rock has been exposed in the major river valleys by glacial processes such as melt water floods (USGS, 1999). Soil orders in the area include Mollisol and Alfisol, which are both silt loams that formed under grassland vegetation.

Physical Stream Characteristics

The River Continuum Concept

The River Continuum Concept is an attempt to generalize and explain longitudinal changes in stream ecosystems (Figure 4.4). The concept proposes a relationship between stream size and progressive shift in structure and functional attributes (Vannote et al., 1980).



Source: Vannote, etal, 1980

FIGURE 4.4
The River Continuum Concept

The conceptual model helps identify the connections between the watershed, floodplain, and the stream system. The concept also describes how biological communities develop and change from headwater areas to the river mouth.

The Continuum Concept hypothesizes that many first to third order headwater streams are shaded by riparian forest canopy. The shading limits algae growth, periphyton, and other aquatic plants. Since energy cannot be created through photosynthesis (autotrophic production), the aquatic community in the stream is dependent on allochthonous materials (materials from outside the channel such as leaves and twigs). Biological communities in the stream are uniquely adapted to the use of externally-derived organic inputs and have, for example, macroinvertebrate communities dominated with shredders and collectors. As we proceed downstream to fourth, fifth, and sixth order streams, the channel widens, which increases available light and levels of primary production. The stream begins to become more dependent on autochthonous materials (material coming from inside the channel). In these downstream sections, species richness of the invertebrate community increases in abundance as they adapt to using both autochthonous and allochthonous food sources.

In large streams and rivers of seventh to twelfth order, there is a trend to increased physical stability, but also a significant shift in structure and biological function. Large rivers develop increased reliance on primary production by phytoplankton. These river sections receive heavy inputs of dissolved and ultra-fine organic particles from upstream. Fine-particle collectors, including zooplankton, dominate invertebrate populations.

The River Continuum Concept is important when interpreting biological community data for the Lower Des Plaines River. The Lower Des Plaines River is a large river system and will not have the characteristics of a headwater stream. Many of the tools used to assess biological integrity of invertebrate and fish communities have been calibrated for headwater streams. Biotic data for large rivers are generally limited. These factors will need to be taken into account when interpreting biological conditions and potential of the Lower Des Plaines River.

Reach-by-Reach Conditions

As outlined in Chapter 1, the Lower Des Plaines River is a waterway that has undergone major physical modification to facilitate the conveyance of treated sanitary waste and commercial navigation. The original stream channel has been relocated, widened, deepened, channelized, and impounded. The Lower Des Plaines River begins at the confluence of the Des Plaines River and the Chicago Sanitary and Ship Canal.

The study area for the Use Attainability Analysis of the Lower Des Plaines River extends from the confluence of the Des Plaines River with the Chicago Ship and Sanitary Canal (CSSC) at the E.J.& E. railroad bridge (River Mile 290.1 near Lockport) downstream to the I-55 Highway Bridge at the River Mile 277.9. The Study area is made up of two distinct impoundment pools. One formed by the Brandon Road Lock and Dam, and one formed by the Dresden Island Lock and Dam. The pools are generally maintained at uniform elevations. The Brandon Road Pool maintained at an average annual elevation of approximately 538.5 feet above sea level (NGVD29). The dam has a head of approximately 34 feet. The Dresden Island Pool is maintained at 505 feet above sea level, and the dam maintains a hydraulic head of approximately 20 feet.

The following is a reach-by-reach narrative description of the Lower Des Plaines River UAA Study area.

Upper Des Plaines River

The Upper Des Plaines River just upstream of the UAA study area is maintained as a natural channel (Figure 4.5). The area is characterized as a large riffle zone with shallow flow and cobble substrate. While outside the study area, the zone is a refuge for organisms that can drift and migrate downstream into the Lower Des Plaines River and repopulate the lower river.

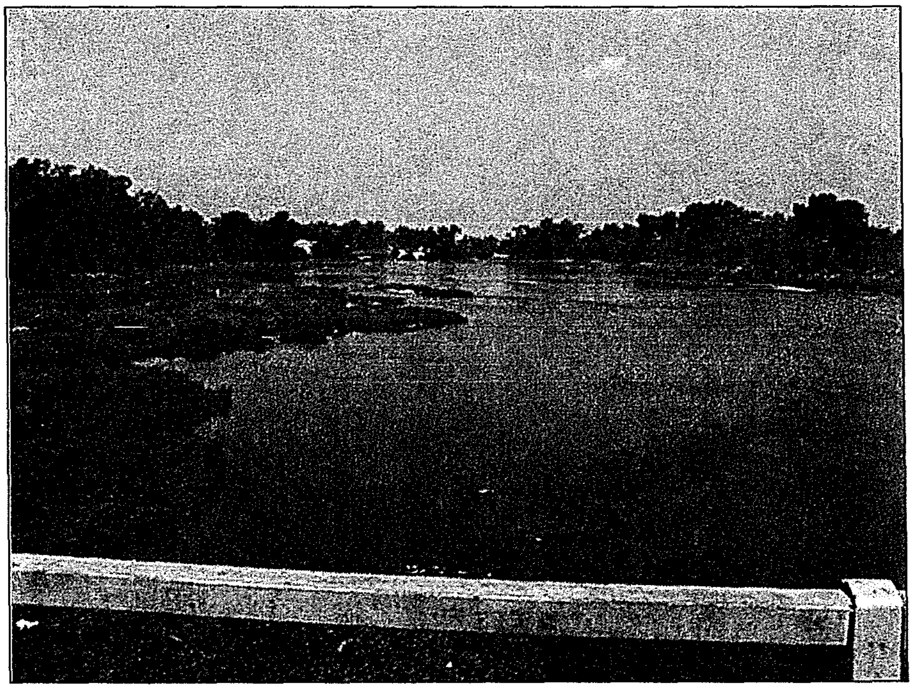


FIGURE 4.5
Upper Des Plaines River Upstream Confluence with Chicago Sanitary and Ship Cannel

Brandon Road Pool

The Brandon Road Pool is a man-made section of the river channel. The river has been deepened and widened to accommodate barge traffic on the river. The walls of the channel have been lined with concrete retaining structures to prevent bank erosion. Figure 4.6 illustrates a typical view of the Brandon Road Pool. Barge traffic in the

Brandon Road Pool consumes a large portion of the river channel (Figure 4.7). Re-suspension of the bottom sediments by the movement of the barges is a common problem in the Brandon Road Pool. The channel is approximately 300 feet wide through much of the pool and the mean depth is approximately 30 feet. Figures 4.8 and 4.9 illustrate typical channel cross-sections.

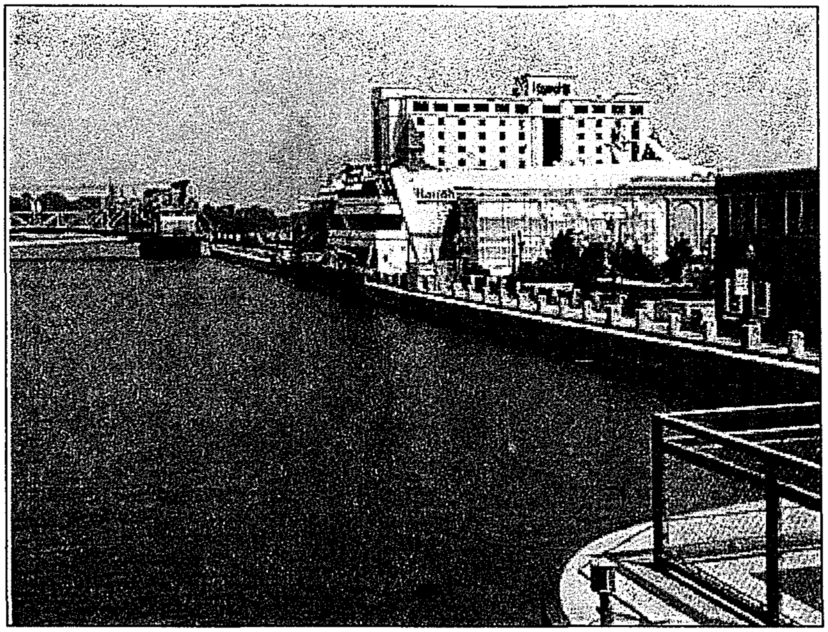


FIGURE 4.6
Typical View of the Brandon Pool, Lower Des Plaines River

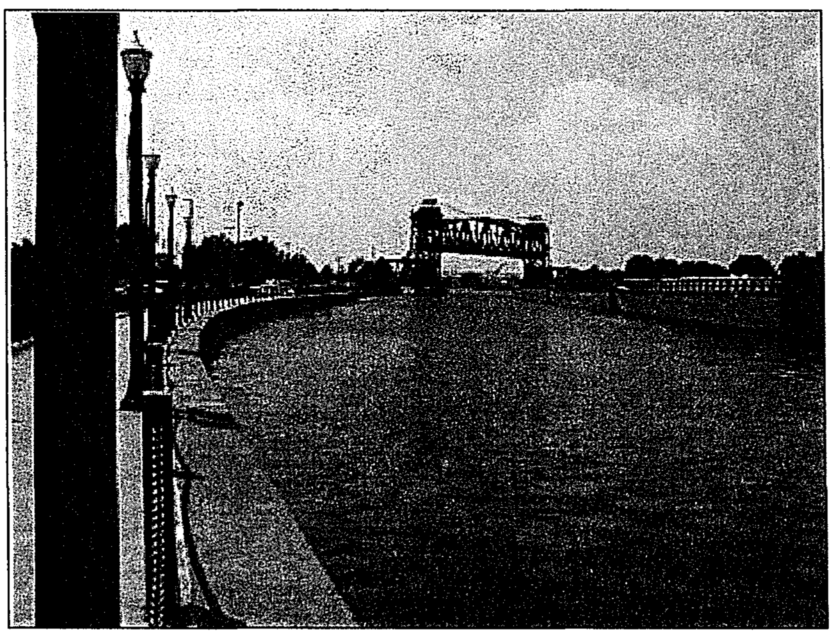


FIGURE 4.7
Barge Traffic on Lower Des Plaines River

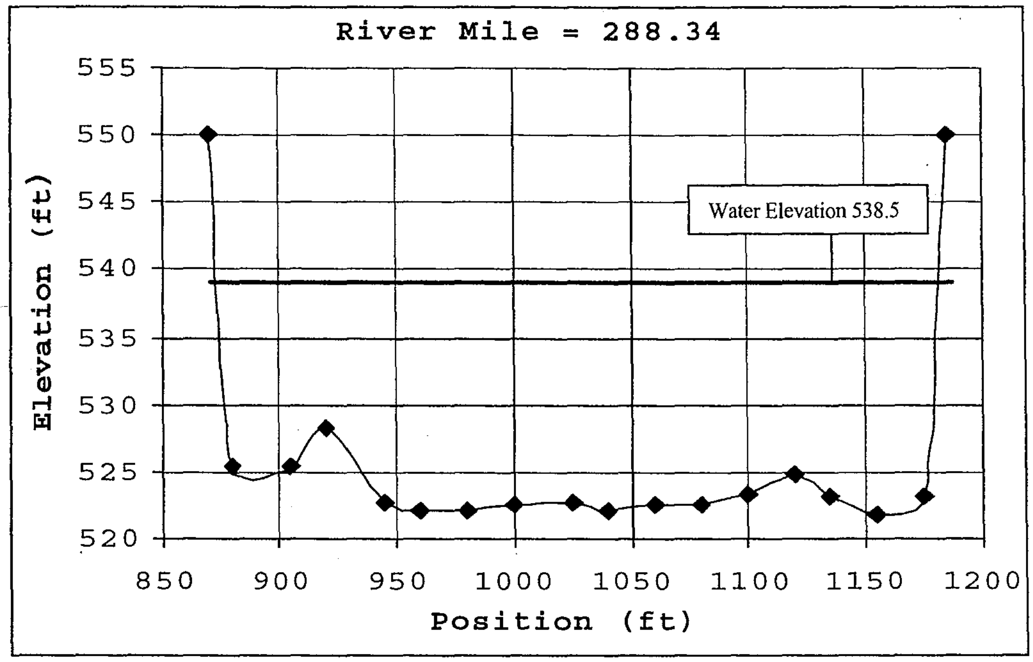


FIGURE 4.8
Brandon Road Pool Channel Cross-Section at River Mile 288.34 (Downtown Joliet) (Source; MWRDGC)

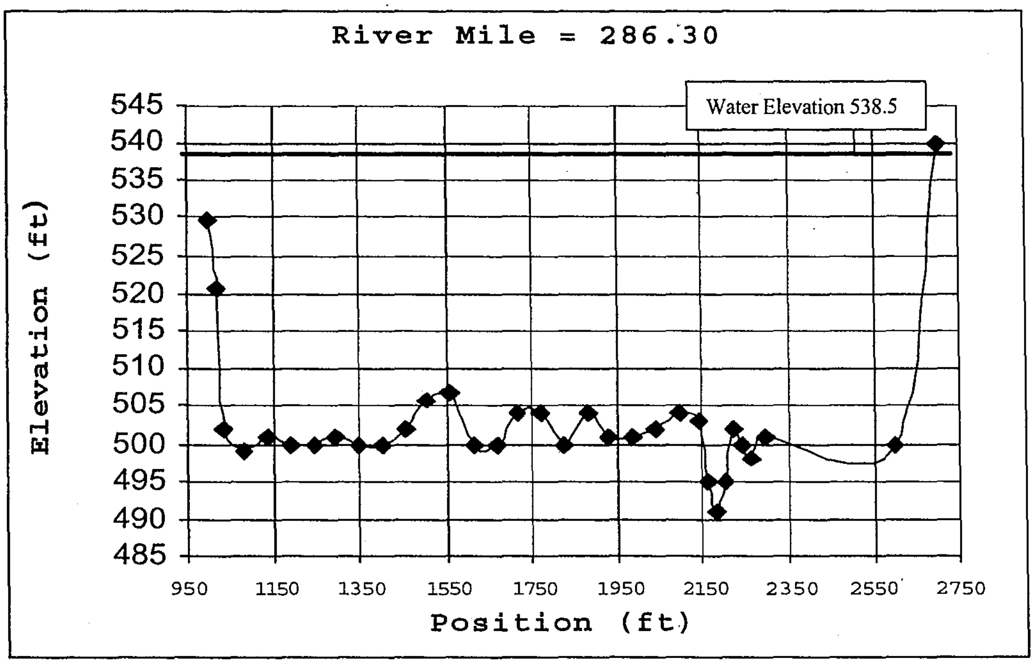


FIGURE 4.9
Brandon Road Pool Channel Cross-Section at River Mile 286.30 (Just Upstream Brandon Road Lock and Dam) (Source; MWRDGC)

Substrate for benthic macroinvertebrates is limited to soft fine-grained organic sediments. Organic detritus and woody debris is limited throughout the pool. Spawning substrate is limited to small cracks and expansion joints in the concrete walls. Shallow substrates and overhanging vegetation do not exist in the pool.

Dresden Island Pool

The Dresden Island Pool extends from the Brandon Road Lock and Dam (River Mile 286) to the Dresden Island Lock and Dam (River Mile 271.5). The UAA study area ends in the middle of the pool at the I-55 Bridge (River Mile 277.9). Below the Brandon Road Lock and Dam, a large tail water riffle zone characterizes the river as illustrated in Figure 4.10.

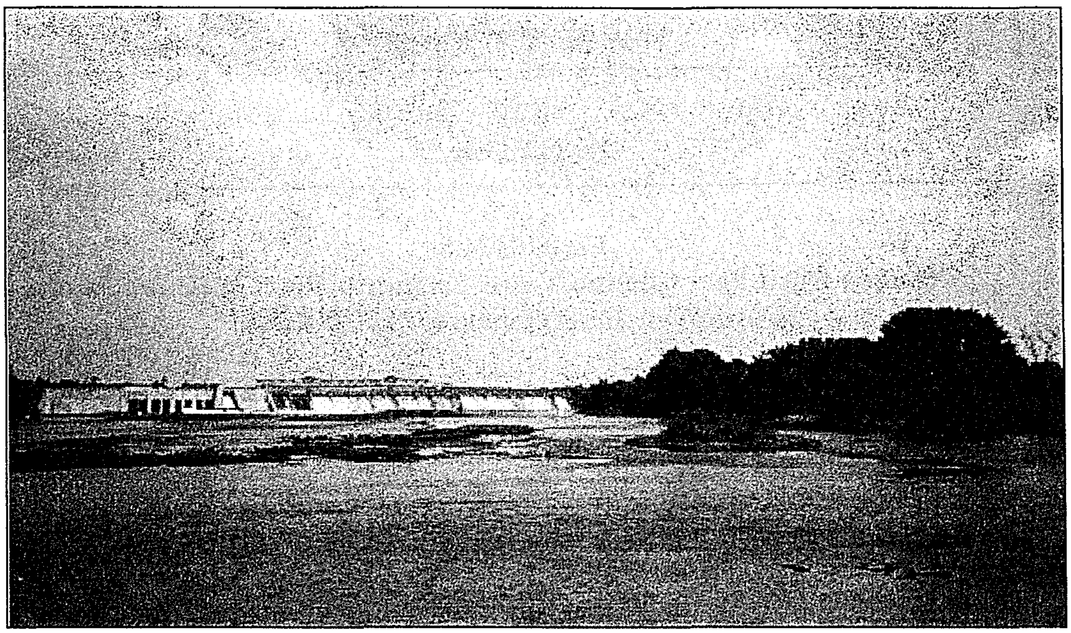


FIGURE 4.10
Area Downstream of Brandon Road Lock and Dam

Unlike the Brandon Road Pool, the banks of the Dresden Island Pool are not armored with concrete walls (Figure 4.11). While the banks are vegetated, the vegetation is indicative of a disturbed community. Riparian vegetation along the banks includes a secondary growth floodplain community of cottonwoods, green ash, elm and various shrubs. Industrial development exists along much of the river as illustrated in Figure 4.12.

Within the Dresden Island Pool, the channel width varies from 500 to 1,500 feet. Within the study reach, there are several backwater areas and tributary mouths that do not exist in the Brandon Road Pool. Maximum depths of the channel are approximately 17 feet in the center of the federal navigation channel. The main channel border is shallow and creates a littoral zone along the bank. Typical channel cross-sections for the pool are illustrated in Figures 4.13 through 4.16.

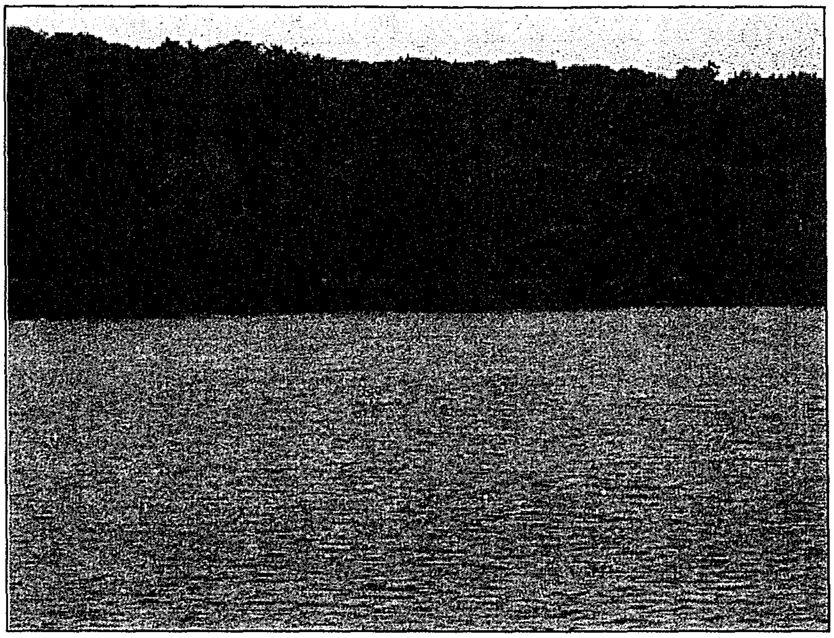


FIGURE 4.11
Representative Stream Bank from River Mile 278.5 to 284.0

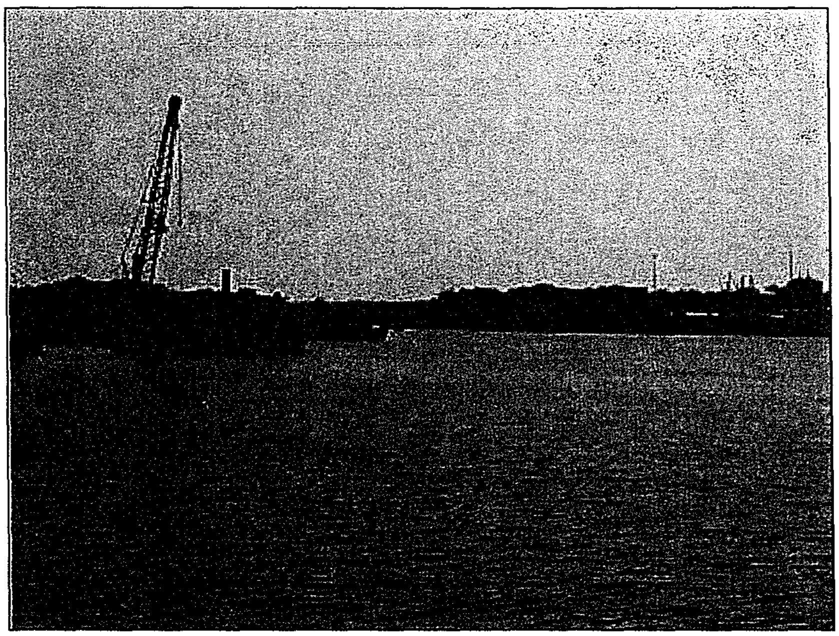


FIGURE 4.12
Industrial Development at River Mile 278.0

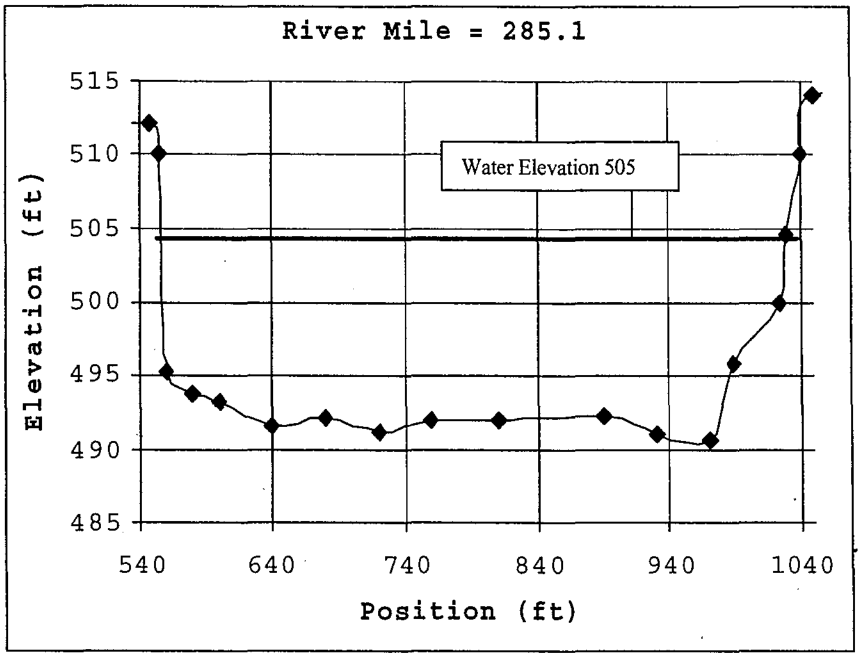


FIGURE 4.13
Dresden Island Pool Cross-Section at River Mile 285.1
(Near Midwest Generation's Joliet Power Plant) (Source; MWRDGC)

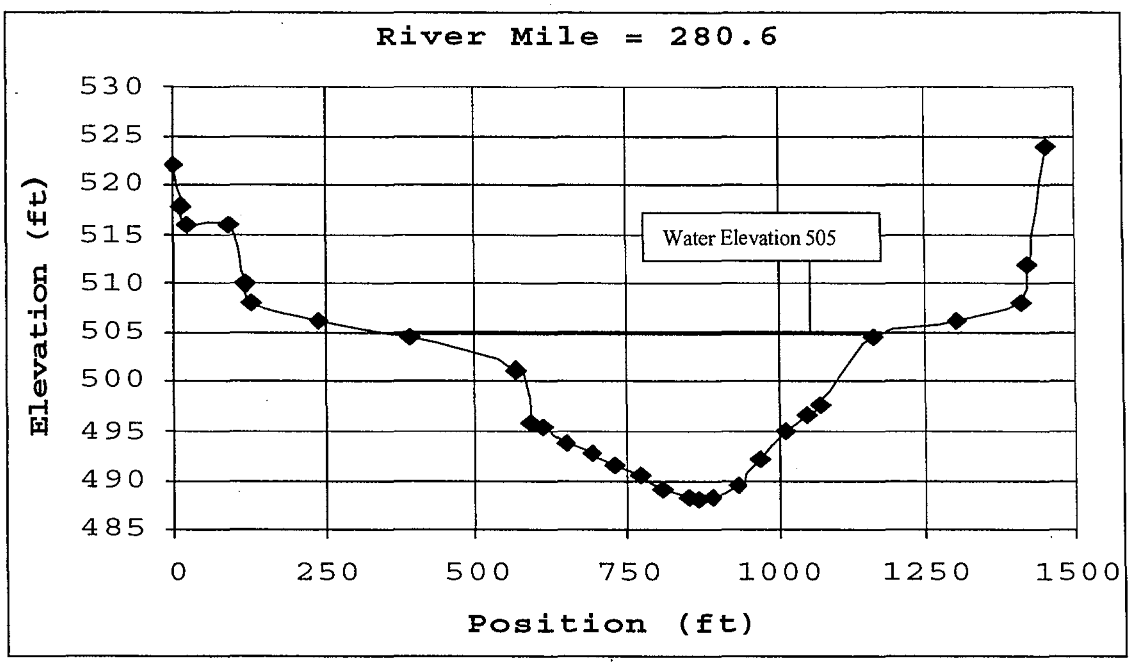


FIGURE 4.14
Dresden Island Pool Cross-Section at River Mile 280.6
(Upstream End of Treats Island) (Source; MWRDGC)

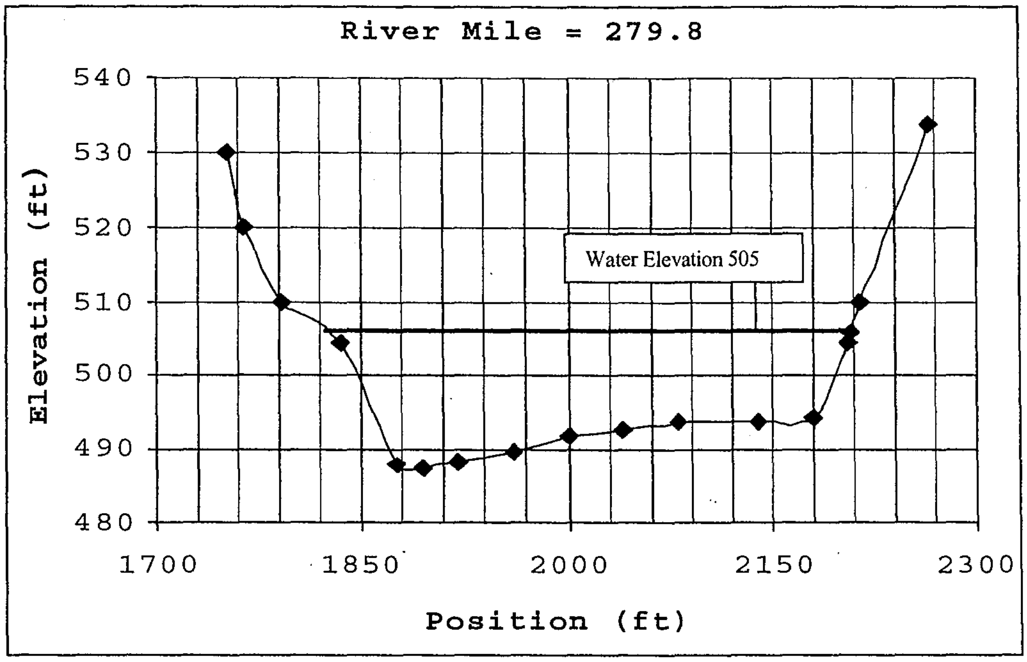


FIGURE 4.15
Dresden Island Pool Cross-Section at River Mile 279.8
(Near Treats Island) (Source; MWRDGC)

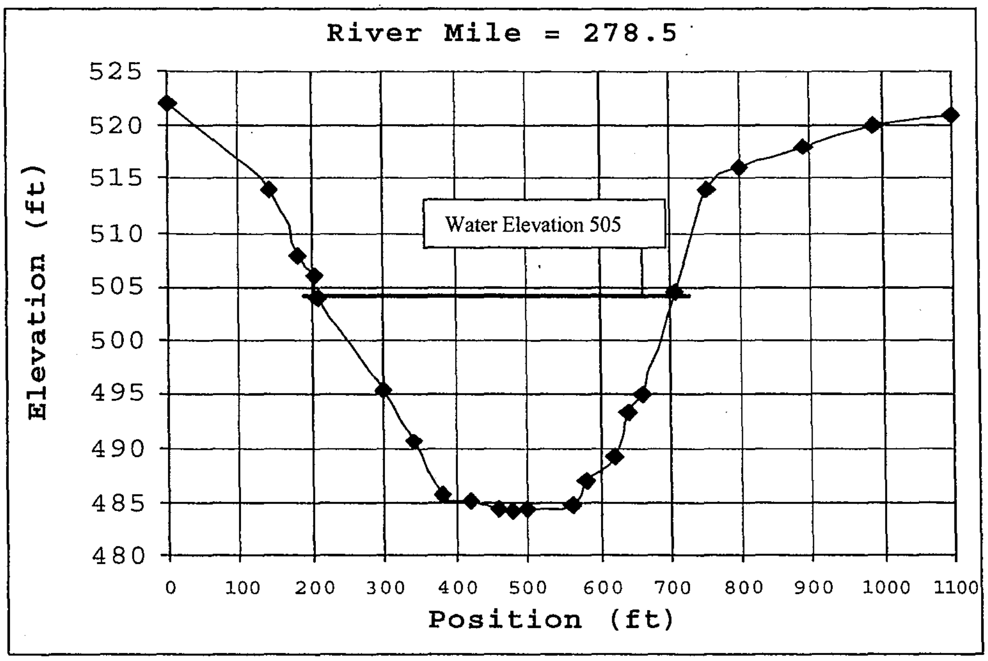


FIGURE 4.16
Dresden Island Pool Cross-Section at River Mile 278.5
(Upstream I-55 Bridge Near Mobil Oil Corp. Joliet Refinery) (Source; MWRDGC)

Habitat Index Values

As outlined in the introduction of this chapter, several authors have proposed methodologies for assessing habitat in streams. The firm, EA Engineering, Science and Technology, on contract with Commonwealth Edison Company, conducted a habitat assessment of the Use Attainability Analysis (UAA) study area in 1993 and 1994 (Commonwealth Edison, 1996). The study used the Qualitative Habitat Evaluation Index (QHEI) developed by the State of Ohio EPA (Rankin, 1989).

Qualitative Habitat Evaluation Index (QHEI) is a macro-scale approach that measures emergent properties of habitat (sinuosity, pool/riffle development) rather than the individual factors that shape these characters (current velocity, depth, substrate size) (Rankin, 1989). The index is used in Ohio to assign water body use based on available habitat. The use is designated by stream segment and not by individual site condition. With this system, one area of poor habitat does not prevent attainment of a high use classification if the majority of the habitat in the water body is good. Conversely, small pockets of good habitat will not allow attainment of a high use if the majority of the habitat is degraded.

While the state of Illinois has a Stream Habitat Assessment Procedure (SHAP), the system is designed predominantly for the assessment of small headwater streams. The OHEI is designed to be used on both headwater (wading) and larger (boatable) streams. Since data was available from the Commonwealth Edison study, it will be used here to qualitatively assess the existing stream habitat. No changes in physical stream habitat have occurred since the Commonwealth Edison was conducted in 1993 and 1994.

OHEI Index System

The following is a description of each of the six QHEI metrics and the individual metric components (Burton and Pitt, 2002). Generally, metrics are scored by checking boxes on the field sheet prepared by the State of Ohio (Figure 4.17). In certain cases, the biologist completing the QHEI sheet may interpret a habitat characteristic as being intermediate between the possible choices; in cases where this is allowed (denoted by the term "Double-Checking"), two boxes may be checked and their scores averaged.

Ohio EPA Site Description Sheet - **QHEI SCORE:**

Stream______________________ RM________ Date__________ River Code____________

Location______________________________ Crew______________________

1) SUBSTRATE (Check *Only* Two Substrate TYPE BOXES. Check all types present): **SUBSTRATE SCORE:**

TYPE Pool Riffle Pool Riffle **SUBSTRATE QUALITY**

[][]-Bldr /Slabs(10)____ ____ [][]-Gravel(7) ____ ____ **Substrate Origin**(Check all) **Silt Cover** (Check One)

[][] -Boulder (9) ____ ____ [][]-Sand(6) ____ ____ []-Limestone(1) []-Riprap (0) []-Silt Heavy(-2)

[][]-Cobble(8) ____ ____ [][]-Bedrock(5) ____ ____ []-Tells(1) []-Hardpan(0) []-Silt Moderate(-1)

[][]-Hardpan(4) ____ ____ [][]-Detritus(3) ____ ____ []-Sandstone(0) []-Silt Normal (0) []-Silt Free(1)

[][]-Muck(2) ____ ____ [][]-Artific(0) ____ ____ []-Shale(-1) **Extent of Embeddness** (Check One)

Total Number of Substrate Types: []>4 (2), []<=4 (0) []-Coal Fines(-2) []-Extensive (-2) []Moderate(-1)

NOTE: (Ignore Sludge that originates from point-sources; score based on natural substances) []-Low(0) []-None(1)

Comments:____

COVER SCORE:

2) INSTREAM COVER

Amount: (Check *ONLY* One or check 2 and AVERAGE)

TYPE (Check *All* That Apply)

[]-Undercut Banks(1) []-Deep Pools(2) []-Oxbows(1) []-Extensive > 75% (11)

[]-Overhanging Vegetation(1) []-Rootwads(1) []-Aquatic Macrophytes(1) []-Moderate 25-75% (7)

[]-Shallows (In Slow Water)(1) []-Boulders(1) []-Logs or Woody Debris(1) []-Sparse 5-25% (3)

[]-Nearly Absent < 5% (1)

Comments:____

3) CHANNEL MORPHOLOGY: (Check *ONLY* One PER Category OR check 2 and AVERAGE) **CHANNEL:**

SINUOSITY	**DEVELOPMENT**	**CHANNELIZATION**	**STABILITY**	**MODIFICATIONS/OTHER**	
[]-High (4)	[]-Excellent (7)	[]-None (6)	[]-High (3)	[]-Snagging	[]-Impound
[]-Moderate(3)	[]-Good (5)	[]-Recovered (4)	[]-Moderate(2)	[]-Relocation	[]-Islands
[]-Low (2)	[]-Fair (3)	[]-Recovering (3)	[]-Low (1)	[]-Canopy Removal	[]-Leveed
[]-None (1)	[]-Poor (1)	[]-Recent or No Recovery(1)		[]-Dredging	[]-Bank Shaping
				[]-One Side Channel Modifications	

Comments:____

RIPARIAN:

4) RIPARIAN ZONE AND BANK EROSION: (Check *ONE* box per bank or check 2 and AVERAGE per bank)

River Right Looking Downstream

RIPARIAN WIDTH	**EROSION/RUNOFF - FLOOD PLAIN QUALITY**		**BANK EROSION**
L R (Per Bank)	L R (Most Predominant Per Bank)	L R (Per Bank)	
[][]-Wide>50m (4)	[][]-Forest, Swamp (3)	[][]-Urban or Industrial(0)	[][]-None or Little (3)
[][]-Moderate10-50m (3)	[][]-Open Pasture/Rowcrop (0)	[][]-Shrub or Old Field(2)	[][]-Moderate (2)
[][]-Narrow 1-5m (2)	[][]-Resid. Park, New Field (1)	[][]-Conserv. Tillage (1)	[][]-Heavy or Severe(1)
[][]-Very Narrow1-5m(1)	[][]-Fenced Pasture (1)	[][]-Mining/construction(0)	
[][]-None (0)			

Comments:____

5) POOL/GLIDE AND RIFFLE/RUN QUALITY **POOL:**

MAX DEPTH (Check 1)	**MORPHOLOGY** (Check 1)	**POOL/RUN/RIFFLE CURRENT VELOCITY** (Check *All* That Apply)	
[]- >1m (6)			
[]- 0.7-1m (4)	[]-Pool Width>Riffle Width(2)	[]-Torrential (-1)	[]-Eddies (1)
[]- 0.4-0.7m (2)	[]-Pool Width=Riffle Width(1)	[]-Fast (1)	[]-Interstitial (-1)
[]- <0.4m (1)	[]-Pool Width<Riffle Width(0)	[]-Moderate (1)	[]-Intermittent (-2)
[]- <0.2m (Pool=0)		[]-Slow (1)	[]-No Pool (0)

Comments:____

RIFFLE:

RIFFLE/RUN DEPTH	**RIFFLE/RUN SUBSTRATE**	**RIFFLE/RUN EMBEDDEDNESS**	
[]-Generally>10cm,Max>50(4)	[]-Stable (e.g. Cobble, Boulder) (2)	[]-Extensive (-1)	
[]-Generally>10cm,Max<50(3)	[]-Mod. Stable (e.g. Pea Gravel)(1)	[]-Moderate (0)	
[]-Generally5-10cm (1)	[]-Unstable (Gravel, Sand) (0)	[]-Low (1)	
[]Generally <5cm (Riffle=0)		[]-None (2)	[]-No Riffle (0)

Comments:____

6) Gradient (feet/mile):______ %Pool:______ %Riffle:______ %Run:______ **GRADIENT:**

FIGURE 4.17

QHEI Field Form

Metric 1: Substrate

This metric includes two components, *substrate type* and *substrate quality.*

Substrate Type – This is the most common substrate type in the stream reach. The field sheet user can check up to two of the appropriate boxes. If one substrate type predominates (greater than approximately 75 to 80% of the bottom area or is clearly the most *functionally* predominant substrate), then this substrate type should be checked twice. A category is provided for artificial substrates. Spaces are provided to note the presence (by check marks or estimates of %, if time allows) of all substrate types present in pools and riffles that each comprises at least 5% of the site (i.e., they occur in sufficient quantity to support species that may commonly be associated with the habitat type). This section must be filled out completely to permit future analyses of this metric. If there are more than four substrate types in the zone that are present in greater than approximately 5% of the sampling area, the investigator checks the appropriate box.

Substrate Quality - *Substrate origin* refers to the "parent" material that the stream substrate is derived from. The investigator checks one box under the substrate origin column unless the parent material is from multiple sources *(e.g.,* limestone and tills). *Embeddedness* is the degree to which cobble, gravel, and boulder substrates are surrounded, impacted in, or covered by fine materials (sand and silt). Substrates should be considered embedded if >50% of surface of the substrates is embedded in fine material. Embedded substrates cannot be easily dislodged. This also includes substrates that are concreted or "armor-plated." Naturally sandy streams are not considered embedded; however, a sand-predominated stream that is the result of anthropogenic activities that have buried the natural coarse substrates is considered embedded. Boxes are checked for *extensiveness* (area of sampling zone) of the embedded substrates as follows: Extensive: >75% of site area, Moderate: 50 to 75%, Sparse: 25 to 50%, Low: <25%.

Silt Cover - the extent to which substrates are covered by a silt layer (i.e., more than 1 inch thick). *Silt Heavy* means that nearly the entire stream bottom is layered with a deep covering of silt. *Moderate* includes extensive coverings of silts, but with some areas of cleaner substrate (e.g., riffles). Normal silt cover includes areas where silt is deposited in small amounts along the stream margin *or* is present as a "dusting" that appears to have little functional significance. If substrates are exceptionally clean, the *Silt Free* box is checked.

Substrate types are defined as:

a. *Bedrock - solid* rock forming a continuous surface.
b. *Boulder - rounded* stones over 250 mm in diameter (10 in.) or large "slabs" more than 256 mm in length *(boulder slabs).*
c. *Cobble - stones* from 64 to 256 mm (2 1/2 to 10 in) in diameter.
d. *Gravel - mixture* of rounded coarse material from 2 to 64 mm (0.8 to 2 1/2 in) in diameter.

e. *Sand - materials* 0.06 to 2.0 mm in diameter, gritty texture when rubbed between fingers.

f. Silt - 0.004 to 0.06 mm in diameter; generally this is fine material, which feels "greasy" when rubbed between fingers.

g. *Hardpan - particles* less than 0.004 mm in diameter, usually clay, which form a dense, gummy surface that is difficult to penetrate.

h. *Marl - calcium* carbonate; usually grayish-white; often contains fragments of mollusc shells.

i. *Detritus* - dead, unconsolidated organic material covering the bottom, which could include sticks, wood, and other partially or undecayed coarse plant material.

j. *Muck-black*, fine, flocculent, completely decomposed organic matter (does not include sewage sludge).

k. *Artificial - substrates* such as rock baskets, gabions, bricks, trash, concrete, etc., placed in the stream for reasons other than habitat mitigation.

Sludge is defined as thick layers of organic matter that is decidedly of human or animal origin. Sludge that originates from point sources is not included in the analysis, and the substrate is based on the underlying material.

Substrate Metric Score - Although the theoretical maximum metric score is > 20, the maximum score allowed for the QHEI is limited to 20 points.

Metric 2: In-Stream Cover

This metric consists of *in-stream cover type* and *in-stream cover amount.* All of the cover types that are present in amounts greater than approximately 5% of the sampling area (i.e., they occur in sufficient quantity to support species that may commonly be associated with the habitat type) should be checked. Cover should not be counted when it is in areas of the stream with insufficient depth (usually < 20 cm) to make it useful. Other cover types with limited utility in shallow water include *undercut banks and overhanging vegetation, boulders, and rootwads.* Under *amount,* one or two boxes may be checked. *Extensive* cover is that which is present throughout the sampling area, generally greater than about 75% of the stream reach. Cover is *moderate* when it occurs over 25 to 75% of the sampling area. Cover is *sparse* when it is present in less than 25% of the stream margins (sparse cover usually exists in one or more isolated patches). Cover is *nearly absent* when no large patch of any type of cover exists anywhere in the sampling area. This situation is usually found in channelized streams or other highly modified reaches (e.g., ship channels). If cover is thought to be intermediate in amount between two categories, the investigator will check two boxes and average their scores. Cover types include: (1) undercut banks, (2) overhanging vegetation, (3) shallows (in slow water), (4) logs or woody debris, (5) deep pools (>70 cm), (6) oxbows, (7) boulders, (8) aquatic macrophytes, and (9) rootwads (tree roots that extend into stream).

Cover Metric Score - Although the theoretical maximum score is >20, the maximum score assigned for the QHEI for the in-stream cover metric is limited to 20 points.

Metric 3: Channel Morphology

This metric emphasizes the quality of the stream channel that relates to the creation and stability of macrohabitat. It includes channel sinuosity (i.e., the degree to which the stream meanders), channel development, channelization, and channel stability. One box under each is checked unless conditions are considered to be intermediate between two categories. In these cases, two boxes are checked and their scores averaged.

a. *Sinuosity* - No sinuosity is a straight channel. Low sinuosity is a channel with only one or two poorly defined outside bends in a sampling reach, or perhaps slight meandering within modified banks. Moderate sinuosity is more than two outside bends, with at least one well defined bend. High sinuosity is more than two or three well-defined outside bends with deep areas outside and shallow areas inside. Sinuosity may be more conceptually described by the ratio of the stream distance between these same two points, taken from a topographic map.
b. *Development* - This refers to the development of riffle/pool complexes. Poor means *riffles* are absent, or if present, shallow with sand and fine gravel substrates; pools, if present, are shallow. Glide habitats, if predominant, receive a Poor rating. Fair means riffles are poorly developed or absent; however, pools are more developed with greater variation in depth. Good means better defined riffles present with larger substrates (gravel, rubble, or boulder); pools vary in depth and there is a distinct transition between pools and riffles. Excellent means development is similar to the Good category except the following characteristics must be present: pools must have a maximum depth of >1 m and deep riffles and runs (>0.5 m) must also be present. In streams sampled with wading methods, a sequence of riffles, runs, and pools must occur more than once in a sampling zone.
c. *Channelization* - This refers to anthropogenic channel modifications. Recovered refers to streams that have been channelized in the past, but which have recovered most of their natural channel characteristics. Recovering refers to channelized streams, which are still in the process of regaining their former, natural characteristics; however, these habitats are still degraded. This category also applies to those streams that were channelized long ago and have a riparian border of mature trees, but still have Poor channel characteristics. Recent or No Recovery refers to streams that were recently channelized or those that show no significant recovery of habitats (e.g., drainage ditches, grass lined or rock riprap banks, etc.). The specific type of habitat modification is also checked in the two columns, but not scored.
d. *Stability* - This refers to channel stability. Artificially stable (concrete) stream channels receive a High score. Even though they are generally a negative influence on fish, the negative effects are related to features other than their stability. Channels with Low stability are usually characterized by fine

substrates in riffles that often change location, have unstable and severely eroding banks, and a high bedload that slowly creeps downstream. Channels with Moderate stability are those that appear to maintain stable riffle/pool and channel characteristics, but which exhibit some symptoms of instability, e.g., high bedload, eroding or false banks, or show the effects of wide fluctuations in water level. Channels with High stability have stable banks and substrates, and little or no erosion and bedload.

e. *Modifications/Other* – This category is checked if impounded, islands present, or levied (these are not included in the QHEI scoring) as well as the appropriate source of habitat modifications.

The maximum QHEI metric score for Channel Morphology is 20 points.

Metric 4: Riparian Zone and Bank Erosion

This metric emphasizes the quality of the riparian buffer zone and quality of the floodplain vegetation. This includes riparian zone width, floodplain quality, and extent of bank erosion. Each of the three components requires scoring the left and right banks (looking downstream). The average of the left and right banks is taken to derive the component value. One box per bank is checked unless conditions are considered to be intermediate between two categories. In these cases, the investigator checks two boxes and averages their scores.

a. *Width of Floodplain Vegetation* - This is the width of the riparian (stream side) vegetation. Width estimates are only done for forest, shrub, swamp, and old-field vegetation. Old-field refers to a fairly mature successional field that has stable, woody plant growth; this generally does not include weedy urban or industrial lots that often still have high runoff potential. Two boxes, one each for the left and right bank (looking downstream), are checked and then averaged.
b. *Floodplain Quality* -The two most predominant floodplain quality types should be checked, one each for the left and right banks (includes urban, residential, etc.), and then averaged. By floodplain we mean the areas immediately outside the riparian zone or greater than 100 feet from the stream, whichever is wider on each side of the stream. These are areas adjacent to the stream corridor and can have direct runoff and erosional effects during normal wet weather.
c. *Bank Erosion* - The following Streambank Soil Alteration Ratings are used;

    1. *None* - streambanks are stable and not being altered by water flows or animals (e.g., livestock) - Score 3.
    2. *Little* - streambanks are stable, but are being lightly altered along the transect line; less than 25% of the streambank is receiving any kind of stress, and if stress is being received it is very light; less than 25% of the streambank is false, broken down, or eroding - Score 3.

3. *Moderate* - streambanks are receiving moderate alteration along the transect line; at least 50% of the streambank is in a natural stable condition; less than 50% of the streambank is false, broken down, or eroding; false banks are rated as altered - Score 2.
4. *Heavy* - streambanks have received major alterations along the transect line; less than 50% of the streambank is in a stable condition; over 50% of the streambank is false, broken down, or eroding - Score 1.
5. *Severe* - streambanks along the transect line are severely altered; less than 25% of the streambank is in a stable condition; over 75% of the streambank is false, broken down, or eroding - Score 1.

False banks mean banks that are no longer adjacent to the normal flow of the channel but have been moved back into the floodplain, most commonly as a result of livestock trampling. The maximum score for Riparian Zone and Erosion metric is 10 points.

Metric 5: Pool/Glide and Riffle-Run Quality

This metric emphasizes the quality of the pool, glide, and/or riffle-run habitats. This includes pool depth, overall diversity of current velocities (in pools *and* riffles), pool morphology, riffle-run depth, riffle-run substrate, and riffle-run substrate quality.

A. POOL/GLIDE QUALITY

1. *Maximum depth of pool or glide - check* one box only (Score 0 to 6). Pools or glides with maximum depths of less than 20 cm are considered to have lost their function and the total *metric is* scored a 0. No other characteristics need be scored in this case.
2. *Current Types - check* each current type that is present in the stream (including riffles and runs; score 2 to 4), definitions are:

   Torrential - extremely turbulent and fast flow with large standing waves; water surface is very broken with no definable, connected surface; usually limited to gorges and dam spillway tailwaters.

   Fast - mostly nonturbulent flow with small standing waves in riffle-run areas; water surface may be partially broken, but there is a visibly connected surface.

   Moderate - nonturbulent flow that is detectable and visible (i.e., floating objects are readily transported downstream); water surface is visibly connected.

   Slow - water flow is perceptible, but very sluggish.

Eddies - small areas of circular current motion usually formed in pools immediately downstream from riffle-run areas.

Interstitial - water flow that is perceptible only in the interstitial spaces between substrate particles in riffle-run areas.

Intermittent-no flow is evident anywhere leaving standing pools that are separated by dry areas.

3. *Morphology - Check Wide* if pools are wider than riffles, *Equal* if pools and riffles are the same width, and *Narrow* if the riffles are wider than the pools (Score 0 to 2). If the morphology varies throughout the site, *average* the types. If the entire stream area (including areas outside of the sampling zone) is pool or riffle, then check riffle = pool.

Although the theoretical maximum score is >12, the maximum score assigned for the QHEI for the Pool Quality metric is limited to 12 points.

B. RIFFLE-RUN QUALITY (score 0 for this metric if no riffles are present)

1. *Riffle/Run Depth - Select* one box that most closely describes the depth characteristics of the riffle (Score 0 to 4). If the riffle is generally less than 5 cm in depth, riffles are considered to have lost their function and the entire riffle metric is scored a 0.
2. *Riffle/Run Substrate Stability - Select* one box from each that best describes the substrate type and stability of the riffle habitats (Score 0 to 2).
3. *Riffle/Run Embeddedness* - Embeddedness is the degree that cobble, gravel, and boulder substrates are surrounded or covered by fine material (sand, silt). We consider substrates embedded if >50% of the surface of the substrates is embedded in fine material, as these substrates cannot be easily dislodged. This also includes substrates that are concreted. Boxes are checked for *extensiveness* (riffle area of sampling zone) with embedded substrates: Extensive: >75% of stream area, Moderate: 50 to 75%, Sparse: 25 to 50%, and Low: <25%.

The maximum score assigned for the QHEI for the Riffle/Run Quality metric is 8 points.

Metric 6: Map Gradient

Local or map gradient is calculated from USGS 7.5 minute topographic maps by measuring the elevation drop through the sampling area. This is done by measuring the stream length between the first contour line upstream and the first contour line downstream of the sampling site and dividing the distance by the change in elevation. If the contour lines are closely "packed," a minimum distance of at least 1 mile is used. Some judgment may need to be exercised in certain anomalous areas (e.g., in the vicinity of waterfalls, impounded areas, etc.), and this can be compared to an in-field, visual estimate, which is recorded on the back of the habitat sheet.

The maximum QHEI metric score for Gradient is 10 points.

Computing the Total QHEI Score

To compute the total QHEI score, add the components of each metric to obtain the metric scores and then sum the metric scores to obtain the total QHEI score. The QHEI metric scores cannot exceed the Metric Maximum Score. The following are the maximum scores assigned to each of the six metrics in the QHEI:

Substrate	20 pts.
In-Stream Cover	20 pts.
Channel Morphology	20 pts.
Riparian Zone	10 pts.
Pool/riffle Quality	20 pts.
Map Gradient	10 pts.
Maximum points	100

The QHEI scores can range from 0 to 100. The meaning of a calculated QHEI value is as follows (Rankin 1989):

> 60 Streams with habitat likely attain warm water habitat use. Use is likely to be consistent with goals of the Clean Water Act.

45 to 60 Streams that may have impaired habitat. Water use designation is determined based on if the stream modifications are reversible or irreversible. Down grading of water use is only done if the stream segment is "irretrievably modified".

< 45 Associates with streams that do not attain warm water habitat biocriteria and have modifications that are generally severe and widespread. These streams are usually given a Modified Warm Water designation.

Results of Commonwealth Edison Company Sampling

The firm EA Engineering, Science and Technology, on contract with Commonwealth Edison Company, conducted a habitat assessment of the Use Attainability Analysis (UAA) study area in 1992 (Commonwealth Edison, 1996). The study results for the Lower Des Plaines River are summarized by habitat type for the Brandon Road Pool in Table 4.3, the Upper Dresden Island Pool in Table 4.4 and Lower Dresden Island Pool in Table 4.5.

TABLE 4.3
QHEI Values for Brandon Road Pool – Lower Des Plaines River

River Mile	Habitat Type[1]				
	MBC	MC	BW	TW	TM
290.0	50.5				
289.3	55.5				
288.9	51.3				
288.7	51.5				
288.0		27			
287.3		37.5			
286.8	38				
286.3	35.5				
286.0	38				
Average	**45.76**	**32.25**			
STDev	**8.23**	**7.42**			

Source: Commonwealth Edison Company (1996)

[1] Habitat Type Description:
MBC - Main Channel Border
MC - Main Channel
BW - Backwater
TW - Tailwater
TM - Tributary Mouth

TABLE 4.4
QHEI Values for Upper Dresden Island Pool – Lower Des Plaines River

	Habitat Type[1]				
River Mile	**MBC**	**MC**	**BW**	**TW**	**TM**
285.7	68				
285.5				69	
285.5	53.3				
285.3				68.75	
285.2	50.75				
285.1	54.75				
285.0		50			
284.9	49.5				
284.8	45.5	43.5			
284.4	47				
274.3	53.5	46.5			
284.2	54.5				
284.0	39.5				
283.8	40				
283.6	43				
283.4	40				
282.9	45				
282.0	44				
281.7	47				
280.6			50.5		
280.5			45.5		
280.0			42		
279.9	56				
279.7	56		57		
279.7		46			
279.3			50.5		
279.0			57		
278.4					51.5
278.3					62
278.2	56				60.5
278.2	49				
277.9			45.5		
Average	**49.62**	**46.50**	**49.71**	**68.88**	**58.00**
STDev	**7.09**	**2.68**	**5.81**	**0.18**	**5.68**

Source: Commonwealth Edison Company (1996)

[1] Habitat Type Description:

MBC - Main Channel Border
MC - Main Channel
BW – Backwater
TW – Tailwater
TM - Tributary Mouth

TABLE 4.5
QHEI Values for Lower Dresden Island Pool – Lower Des Plaines River

River Mile	Habitat Type[1]				
	MBC	MC	BW	TW	TM
277.6	50		42.5		
277.4			49		
277.2			50		
276.9					46
276.5	51.5				
276.2			44		
276.1	46.5				
276.0			48		
275.9			48		
275.5			48		
274.8					54.8
274.4	60				
273.7	47				
273.5	45.5				
273.0	45				
272.9	58				
272.8	50.5				
272.4	54.5				
272.1	58.9				
272.0	59				
271.9	53				
271.7	44				
Average	**51.67**		**47.07**		**50.40**
STDev	**5.68**		**2.75**		**6.22**

Source: Commonwealth Edison Company (1996)

[1] Habitat Type Description:
- MBC - Main Channel Border
- MC - Main Channel
- BW - Backwater
- TW - Tailwater
- TM - Tributary Mouth

Figure 4.18 illustrates the accumulated QHEI values for the various reaches in the Lower Des Plaines River study area. Figure 4.19 provides a legend for reading box and whisker plots. The Brandon Road pool has an accumulative medium value of 37 and mean value of 42, indicating stream modifications that are generally severe and widespread, and conditions that do not provide habitat to support full warm water use. The Upper and Lower Dresdén Island pool both have accumulative medium values of 49 and mean values of 50, indicating less than optimum habitat that, if irreversible, could justify a modification of stream use classification under the Ohio EPA stream classification system.

Figure 4.20 illustrates trends in the QHEI values by stream reach and habitat type. From the data we see that much of the quality habitat in the Lowers Des Plaines River exists at tailwater areas below the dams and at tributary mouths. The main channel and main channel boarders provide marginal habitat, with QHEI scores typically less than 50.

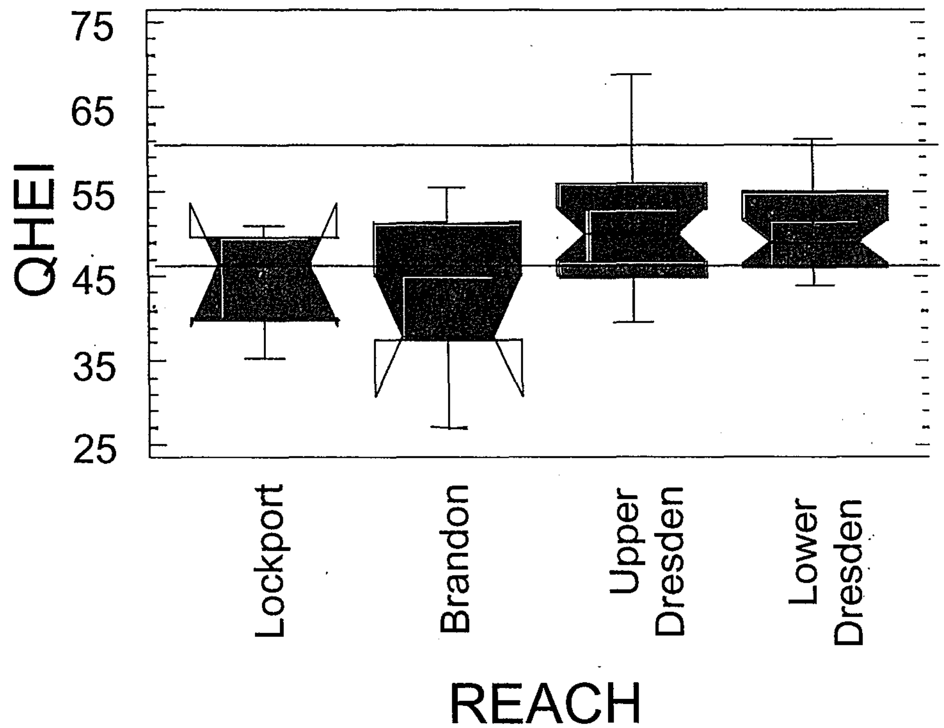


FIGURE 4.18
Whisper Plot of Accumulated QHEI Values by Stream Reach for the Lower Des Plaines River

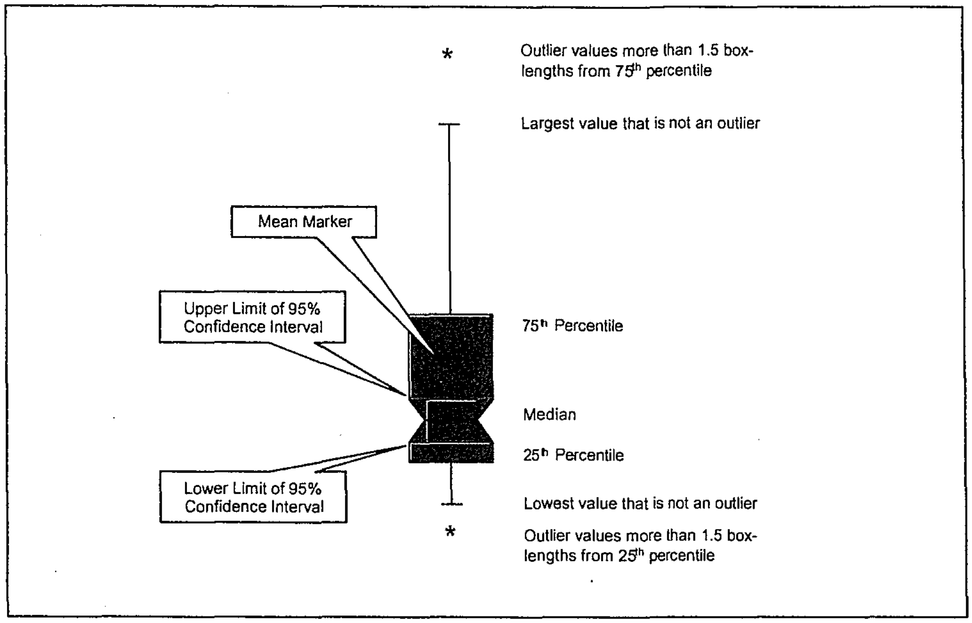


FIGURE 4-19
Box and Whisker Plot Legend

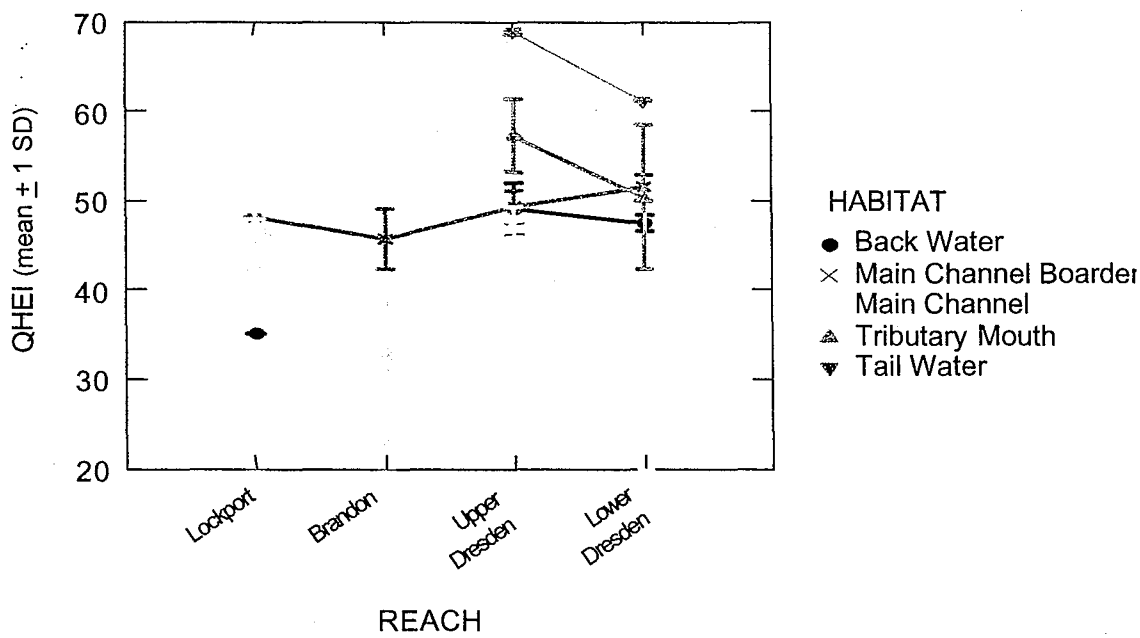


FIGURE 4.20
Trends in QHEI Values by Stream Reach and Habitat Type

A statical analysis of variance, summarized in Table 4.6, illustrates that QHEI scores are controlled predominately by specific habitat types. The Dresden Island pool scores better than the Brandon Road pool predominantly due to the presence of tailwater and tributary mouth habitats. Poor habitat scores throughout the Lower Des Plaines River are due to the following reasons:

- lack of riffle/run habitat
- limited hard substrates (i.e. gravel/cobbles)
- channelization
- poor riparian habitat
- lack of in-stream cover
- impounded water

TABLE 4.6
Analysis of QHEI Variance for Habitat Types by Reach
(Type III Sum of Squares)

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
Reach	373.46	3	124.487	3.03	0.0356
Habitat	1051.86	4	262.965	6.40	0.0002
Residual	2670.60	65	41.086	-	-
Total (corrected)	4344.89	72	-	-	-

Source: Dr. Tim Elhinger

Irreversible Nature of Habitat Alterations

Box 4.1 (1.1) outlines the six reasons for a change of the designated use of a water body as outlined in Federal Regulation 40 CFR 131. The UAA task is not only to assess the current situation but also to find out whether or not the designated use is attainable. Reasons 1, 2 and 3 do not relate to physical habitat and will not be discussed in this chapter. Reasons 4, 5 and 6 will be reviewed to determine if the circumstances in the Lower Des Plaines River meet these conditions and if the circumstances are reversible or irreversible and the use is attainable.

To understand if the habitat alterations that result in the less than optimum QHEI scores are correctable, we need to first understand which habitat features are controlling the scores. Table 4.7 summarizes the individual metric scores for a representative group of QHEI sample locations. As seen in the table several metric scores cannot be changed without major physical alterations to the stream channel and removal of the lock and dam system that forms the two impoundments in the Lower Des Plaines River.

(This Table will be referred as Box 1.1 and eventually taken out from this chapter.)

Box 4.1 Six reasons for a change of the designated use and/or water quality standards of a water body (40 CFR 131)

(1) Naturally occurring pollutant concentrations prevent attainment of the use; or

(2) Natural, ephemeral, intermittent or low flow or water levels prevent the attainment of the use unless these conditions may be compensated for by the discharge of a sufficient volume of effluent discharge without violating State conservation requirements to enable uses to be met; or

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place; or

(4) Dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use; or

(5) Physical conditions related to the natural features of the water body, such as the lack of proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or

(6) Controls more stringent that those required by Sections 301(b)(1)(A) and (B) and 306 of the Act would result in substantial and wide-spread adverse social and economic impact.

TABLE 7.7
QHEI Metric Scores for Selected Sites in the Lower Des Plaines River

River Mile	Substrate	Cover	Channel	Riparian	Pool	Riffle	Gradient	QHEI
				Brandon Road Pool				
290.5	16	7	7.5	6	9	0	6	51.5
289.3	17	7	9	7.5	9	0	6	55.5
288.9	16	8	7.5	4	9	0	6	50.5
286.8	1	14	6	3	8	0	6	38.0
Average	**12.5**	**9.0**	**7.5**	**5.1**	**8.75**	**0**	**6.0**	**48.9**
				Dresden Island Pool				
285.5	16	13	9	3.5	9	0	6	56.5
284.6	12	5	9	6.5	10	0	6	48.5
284.6	1	9	9	7	8	0	6	40.0
276.5	10	10	9	7.5	9	0	6	51.5
274.4	16	12	9	8	9	0	6	60.0
272.8	10	9	10	7.5	8	0	6	50.5
272.1	18	17	6	6	10	0	6	63.0
271.9	9	12	9	6	8	0	6	50.0
Average	**11.5**	**10.9**	**8.75**	**6.5**	**8.9**	**0**	**6.0**	**52.5**
Max. Score	**20**	**20**	**20**	**10**	**10**	**10**	**10**	**100**

Source: Commonwealth Edison Company (1996)

The stream channelization, lock and dam system, and routine dredging needed to maintain the federal navigation channel plays a major role in affecting the habitat in the Lower Des Plaines River. QHEI scores for following metrics are controlled by the navigation system:

substrate (lack of coarse materials such as gravel of boulders)
channel morphology (lack of sinuosity and channel development)
pool quality (much of the river is in deep pool)
riffle quality (no riffle habitats present)
stream gradient (gradient controlled by local dams)

Scores for these categories cannot be improved without removal or major modification to the navigation system. In Federal Regulation 40 CFR 131, navigation is listed as a "typical" and protected use. As long as commercial navigation takes place on the Lower Des Plaines River, changes to the above habitat features are irreversible. Impoundment of the river by the Brandon Road and Dresden Island Lock and Dams creates a deep pool environment that is lacking in coarse substrate, channel diversity, riffle habitat, and gradient. The physical habitat formed by the navigation system fall under reasons 4 and 5 for a change of the designated use outlined in Box 1.1.

Commercial navigation is a multimillion -dollar industry of the Lower Des Plaines River. The Upper Illinois and Chicago waterway system represents a major navigational connection between the Great Lakes (Atlantic Ocean, grain producing states along the Illinois and Mississippi River and Gulf of Mexico) Elimination of commercial barge traffic could cause "wide-spread adverse social and economic impact" and trigger reason number 6 outlined in Box 1.1.

Two habitat categories as measured with the QHEI could be improved through artificial management. The categories are in-stream cover and riparian zone and bank erosion. As seen in Table 4.7 in-stream cover values in both the Brandon Road and Dresden Island Pool are about half of the potential maximum value of 20. Placement of artificial in-stream habitat could improve the habitat scores. However, due to the depth of the water maintained for the navigation channel and routine barge traffic on the river, in-stream habitat improvement opportunities would be limited to the boarders of the stream channel. In the Brandon Road Pool because of the concrete and sheet pile retaining walls, the opportunities for in-stream habitat improvement are minimal or non existent. At best, in-stream habitat features placed in the Brandon Road Pool could raise the QHEI scores only 3 points. Greater opportunities exist in the channel boarder areas of the Dresden Island Pools, which could allow QHEI scores to improve 6 to 7 points.

Riparian zone metric scores are also below the maximum potential of 10 points. Increasing the width of the riparian buffers along the stream could improve habitat scores. Due to the retaining walls along the Brandon Road Pool, downtown Joliet development and the fact that the water level in the pool is above the downtown elevation, there is almost no riparian buffer of the stream channel; therefore, improvements in this stream reach would have limited benefits to in-stream organisms. Potential improvements in riparian buffer areas could potentially increase QHEI values by 3 to 4 points.

The addition of in-stream cover and better riparian buffers are better in the Dresden Island pool and improvements along the stream channel could potentially increase the QHEI scores for the Dresden Island Pool to above the score of 60 used by Ohio EPA to define warm water habitat. Modifications to the Brandon Road Pool would improve the QHEI scores, however unlikely enough to reach values above 50.

Conclusion

Habitat throughout the Lower Des Plaines River is degraded due to channelization and impoundment of the river. QHEI scores for the study area are below the recommended value of 60 used by Ohio EPA to define warm water habitat use that is consistent with goals of the Clean Water Act. Habitat scores in the Brandon Road pool (medium QHEI value of 37) indicates stream modifications that are generally severe, irreversible and widespread, and conditions that do not provide habitat to support full warm water use. While the Dresden Island pool has higher habitat index scores, the current values still indicate a system that does not meet the optimum for warm water use.

Poor habitat in the Lower Des Plaines River is the result of a lack of riffle/run habitat, limited hard substrates (i.e. gravel/cobbles), channelization, poor riparian habitat, lack of in-stream cover, and impounded water. The above factors are the result of the channelization and impoundment of the river for maintenance of the Upper Illinois River Waterway of which the Des Plaines River is a part. At the current time, the river is heavily used for commercial barge traffic, a protected use under the Clean Water Act. While commercial barge traffic continues on the Lower Des Plaines River, the major causes of the degraded habitat are considered irreversible, as the bck and dam system is vital to commercial navigation. Artificial placement of in-stream cover and improvements in riparian buffer areas could improve habitat quality.

In the Dresden Island Pool improvements in in-stream cover and riparian buffers could potentially improve QHEI scores to above the recommended Ohio value of 60. Artificial habitat improvements in the Brandon Road Pool could improve QHEI scores, however unlikely much above 50, resulting in habitat that would still not meet full warm water use as defined by Illinois General Use designation.

Two habitat categories as measured with the QHEI could be improved through artificial management. The categories are "in-stream cover" and "riparian zone and bank erosion". Placement of artificial in-stream and riparian corridor habitat could improve the habitat scores. In-stream habitat would include undercut banks, over hanging vegetation, boulders and rootwads. Expansion of the vegetative corridor along both stream banks could improve the riparian zone scores. The addition of woody vegetation along the corridor would provide additional habitat for macroinvertebrates and cover for fish. However, due to the depth of the water maintained for the navigation channel and routine barge traffic on the river, in-stream habitat improvement opportunities would be limited to the boarders of the stream channel. In the Brandon Road Pool because of the concrete and sheet pile retaining walls, the opportunities for in-stream habitat improvement are minimal or non existent.

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CHAPTER 5

EXISTING AND POTENTIAL MACROINVERTEBRATE COMMUNITY

Introduction

Benthic macroinvertebrates are an important component of a balanced ecosystem and have long been used as indicators of ecological health of streams. The group is operationally defined as those invertebrates retained on sieve mesh sizes greater than 0.2 mm (Hynes, 1970), however larger size sieves of 0.5 or 0.95 mm (U.S. Standard No. 30) are routinely used (EPA, 1989c.). The functional feeding groups include herbivores, omnivores, and carnivores. The group is made up of deposit and detritus feeders, collectors, shredders, and grazers.

The benthic macroinvertebrate community has been used for many years to qualitatively and more recently (in the United States), quantitatively assess water quality and pollution problems. Benthic invertebrates have been used for quantitative pollution assessment in Europe for almost one hundred years (Kolkwitz and Marson, 1908). The advantages of using macrobenthos in water quality assessments are outlined in Table 5.1.

For the purposes of this Use Attainability Analysis, macroinvertebrate will be used to deter mine if the current stream conditions are meeting the goals of "biological integrity" as defined in Section 101 of the Clean Water Act (CWA). Biological integrity in the State of Illinois is defined by the state's "General Use Standards". If the stream does not meet the General Use Standards, the reasons outline in Box 1.1 (Chapter 1) can be used as justification for a change in designated use.

There is a wealth of reference information available to assist in the use of macroinvertebrates as monitoring tools, including Armitage (1978), Benke et al. (1984), Brinkhurst (1974), Cairns (1979), Cummins et al. (1984), Cummins and Wilzbach (1985), Edmondson and Winberg (1971), Goodnight and Whitley (1960), Hart and Fuller (1974), Hellawell (1978, 1986), Hilsenhoff (1977), Howmiller and Scott (1977), Hynes (1960, 1970), Holme and McIntyre (1971), Hulings and Gray (1971), Lenat (1983), Lind (1985), Merritt and Cummins (1984), Mason (1981), Metcalfe (1989), Milbrink (1983), Meyer (1990), Neuswanger et al. (1982), Pennak (1989), Posey (1990), Resh (1979), Resh and Rosenberg (1984), Resh and Unzicker (1975), Reynoldson et al. (1989), Ward and Stanford (1979), Warren (1971), Waters (1977), Welch (1948), Welch (1980), Winner et al. (1975), EPA (1989a,c, 1990a,c, 1999), and OEPA (1989).

Sampling of the benthic macroinvertebrate community can be done either through sampling the bottom substrate or by establishing artificial substrates for colonization. Sampling of the bottom substrate is done through either collecting grab samples of bottom sediment or by disturbing the streambed and collecting dislodged organisms in a fine mesh net. In a large stream, sediments are usually collected with a sampling dredge such as a Ponar or Ekman grab sampler (Elliott and Drake, 1981). Artificial substrates are used to measure drifting organisms that colonize on the sample device. Artificial samplers remove the substrate variable and provide known sampling areas and exposure times. The Hester-Dendy sampler is one of the more common artificial substrates used. Unfortunately, there are some disadvantages that include: some taxa may not utilize the substrate, substrates are colonized primarily by upstream drift organisms, and effects from contact with possibly contaminated sediments is reduced or eliminated. Sample results can be influenced by the sampling device used and the technique needs to be taken into account when evaluating the data.

TABLE 5.1
Advantages of Using Macroinvertebrates in the Evaluation of Biotic Integrity

Macroinvertebrate assemblages are good indicators of localized conditions. Because many benthic macroinvertebrates have limited migration patterns or a sessile mode of life, they are particularly well-suited for assessing site-specific impacts (upstream-downstream studies).
Macroinvertebrates integrate the effects of short-term environmental variation. Most species have complex life cycles of approximately one year or more. Sensitive life stages will respond quickly to stress: the overall community will respond more slowly.
Degraded conditions can often be detected by an experienced biologist with only a cursory examination of the benthic macroinvertebrate assemblage. Macroinvertebrates are relatively easy to identify to family; most "intolerant" taxa can be identified to lower taxonomic levels with ease.
Benthic macroinvertebrate assemblages are made up of species that constitute a broad range of trophic levels and pollution tolerances, thus providing strong information for interpreting cumulative effects.
Sampling is relatively easy, requires few people and inexpensive gear, and has minimal detrimental effect on the resident biota.
Benthic macroinvertebrates serve as a primary food source for fish, including many recreationally and commercially important species.

Source: EPA, 1999

Many tools have been established to attempt to interpret the meaning of benthic macroinvertebrate data. Tools have ranges from single metric analysis--such as pollution tolerance used in Illinois's Macroinvertebrate Biotic Index (MBI), to multi-metric indexes such as Ohio's Invertebrate Community Index (ICI). The Biological Sub-Committee of the Lower Des Plaines River Use Attainability Analysis (UAA) Workgroup proposed two approaches to the analysis of collected data. The first approach involves evaluation of several individual metrics. The second involves using biological indexes to understand the entire community structure.

Historic Data

Benthic macroinvertebrates were sampled by Illinois Water Survey in the early 1970s (Butts, 1974; Butts et al., 1975). The sediments and benthic communities during that time were very different from present. The sediments had mostly an oily, musty consistency. The only organisms found were sludgeworms from the tubificadae family (most likely *Limnodrilus hoffmeisteri*) and bloodworms (*Chironomus* larvae). The former occurred in massive quantities in the Brandon Road and Dresden Island pools. The number of worms in the samples above mile 281.4 was so great that field picking and counting was almost impossible (Butts et al., 1975). The numbers of invertebrates in the pools were estimated as ranging from 4,000/m^2 at mile 280.6 to 30,000/m^2 at mile 281.4.

The presence of the large quantities of tubificidae worms thirty years ago has some ramification on evaluation of toxicity. These organisms thrive on organic content of sediment and are highly tolerant of organic pollution (see also Chapter 3). According to USEPA (1994) the tubificid *Limnodrilus hoffmeisteri* is considered tolerant of metal contamination. *Chironomus attenuatus* is listed as tolerant to heavy metals, but *C. riparius* is listed as sensitive to heavy metals (Klemm, et al, 1990). Twelve other chronomus species listed by Klemm et al. (1990) are not identified as tolerant or sensitive.

Sampling conducted by Illinois Environmental Protection Agency (IEPA) at river mile (RM) 288.7 in the Brandon Pool in the late 1980's and early 1990's found Turbellaria, the midge *Nanodadius sp.*, and the worms *Dero sp.* and Nias *variabilis* to dominate on Hester-Dendies samplers. Samples collected in the Dresden Pool at RM 273.5 and 278.0 had greater species richness and were dominated by chironomids such as *Naocladius distinctus* and *Polypedilum convictum*, and caddisflies such as *Hydropsyche sp.* at RM 278.0 (I-55) and *Cyrnellus fraternus* at RM 273.5 (Bay Hill Marina) (Commonwealth Edison Company, 1996).

Commonwealth Edison Company collected Macroinvertbrate data in 1993 and 1994 as part of an assessment of the Upper Illinois Waterway required by a variance issued to the company's Joliet, Will County, Crawford and Fisk power plants. The data collected using Hester-Dendy artificial samplers was synthesized into a series of biotic index values using the Ohio Invertebrate Community Index (ICI). The results of the study are summarized in Table 5.2. For comparison the ICI values have the following meanings:

Category	Score Range
Exceptional	44 - 54
Good	32 – 42
Fair	12 – 30
Poor	2 – 12
Very Poor	0

TABLE 5.2
Commonwealth Edison ICI Scores for Macroinvertebrates Collected in 1993 and 1994

River Mile	ICI
273.5	16
275.0	14
276.2	16
276.9	16
277.6	14
277.6	12
277.9	12
279.3	12
279.9	18
284.3	14
285.3	20
286.0	10
287.3	22
288.9	22

Source: Commonwealth Edison Company, 1996

Summary of Current Data from MWRGC and IEPA

To determine the current condition of the Lower Des Plaines River it was the recommendation of the Lower Des Plaines River Workgroup - Biological Subcommittee that only data from the past five years be used in the analysis. Benthic macroinvertebrate data used for this UAA comes from several sources: the Metropolitan Water Reclamation District of Greater Chicago (MWRGC) and the Illinois Environmental Protection Agency (IEPA). Two collection methods were used to obtain macroinvertebrate information--artificial substrates (Hester-Dendy, HD, multiplate samplers) and Ponar dredge samples of natural substrate material (generally consolidated soft sediment). The use of two different collection methods can be attributed to the type and variability of habitat and physical changes in river channel morphology. The data is limited to samples collected during the summer of 2000.

Data was collected in the stream reaches outlined in Table 5.3. The study reaches for the Lower Des Plaines River Use Attainability Analysis are from the I-55 Bridge (River Mile 277.8) to the confluence of the Des Plaines River and the Chicago Sanitary Ship Canal (River Mile 290.0). Areas in the Lockport and Lower Dresden Pools, outside the study area, are presented for reference purposes.

TABLE 5.3
Stream Reaches with Available Benthic Macroinvertebrate Data

Pool	Stream Reach	River Miles
Lockport Pool	Upstream of Lockport Lock and Dam	Upstream of Mile 291
Brandon Pool	Brandon Road Lock and Dam upstream to Lockport Lock and Dam	Miles 286 to 291
Upper Dresden Pool	I-55 Bridge upstream to Brandon Road Lock and Dam	Miles 277.8 to 286
Lower Dresden Pool	Below I55 Bridge downstream to Dresden Lock and Dam	Miles 271.5 to 277.8

Trends in Macroinvertebrate Data

Temporal data was limited; therefore, no long-term trends were possible in this analysis. In addition, the data was further narrowed to one index period (August) and the influence of other large river inputs removed (Kankakee River), to assist in reducing the variability associated with the data set. Conclusions drawn from this small data set should be done judiciously.

A set of community characteristics (metrics) was selected to look at their response spatially throughout the UAA reach. These characteristics were selected based on literature and consensus among biologists from IEPA, MWRGC, USEPA, and private consultants.

Evaluation of Community Characteristics (Metrics)

Assessment of biological condition (integrity) may be indicated by evaluation of community characteristics (metrics). Ideally each metric chosen will measure a different characteristic of the community structure and have a different range of sensitivity to stressors. The metrics outlined in Table 5.4 were selected for evaluation. The data for each metric is presented in Appendix E, and illustrated by showing the individual values by river mile and grouped in the form of whisper plots for the study reaches outlined Table 5.3.

TABLE 5.4

Individual Macroinvertebrate Metric Used in Analysis

Metric	Definition	HD	Ponar Grab	Predicted Response to Increasing Perturbation
Total Number of Taxa	Measures the overall variety of the macroinvertebrate assemblage	X	X	**Decrease** (DeShone 1995, Barbour et al. 1996, Fore et al. 1997, Voshell 1997)
Number of EPT Taxa	Number of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)	X	X	**Decrease** (DeShone 1995, Barbour et al. 1996, Fore et al. 1997, Voshell 1997)
% EPT Taxa	Percent composition of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)	X	X	**Decrease** (DeShone 1995, Barbour et al. 1996, Fore et al. 1997, Voshell 1997)
Number of Intolerant Taxa	Taxa richness of those organisms considered sensitive to perturbation	X	X	**Decrease** (DeShone 1995, Barbour et al. 1996, Fore et al. 1997, Voshell 1997)
% Tolerant Individuals	Percent of macrobenthos considered to be tolerant of various types of perturbation	X	X	**Increase** (DeShone 1995, Barbour et al. 1996, Fore et al. 1997, Voshell 1997)
Number of Taxa in Family Chironomidae	Number of taxa of chironomid (midge) larvae	X		**Decrease** (Hayslip 1993, Barbour et al. 1996)
% Chironomidae	Percent of midge larvae	X	X	**Increase** (Barbour et al. 1994)
% Chironominae	Percent of midge larvae from the subfamily Chironominae	X	X	**Undocumented**
% Orthocladinae	Percent of midge larvae from the subfamily Orthocladinae	X	X	**Increase** (Kerans and Karr 1994, Fore et al. 1996, Barbour et al. 1996)
% Tanypodinae	Percent of midge larvae from the subfamily Tanypodinae	X	X	**Undocumented**
% Tribe Tanytarsini	Percent of midge larvae from the tribe Tanytarsini	X		**Decrease** (DeShone 1995)
% Oligochaeta	Percent of aquatic worms	X	X	**Elevated under organic enrichment** (Kerans and Karr 1994)
% Hydropsychidae	Percent of caddisfly larvae from the family Hydropsychidae to Total Trichoptera	X		**Increase** (Kerans and Karr 1994, Fore et al. 1996, Barbour et al. 1996)

Metric	Definition	HD	Ponar Grab	Predicted Response to Increasing Perturbation
% Mollusca	Percent of snails and bivalves	X	X	**Decrease** (Kerans and Karr 1994, Fore et al. 1996, Barbour et al. 1996)
%Isopoda	Percent of isopods	X	X	**Increase** (Kerans and Karr 1994, Fore et al. 1996, Barbour et al. 1996)
% Amphipoda	Relative abundance of scuds	X	X	**Decrease** (Kerans and Karr 1994, Fore et al. 1996, Barbour et al. 1996)
% Odonata	Percent of dragonfly and damselfly nymphs	X	X	**Increase** (Kerans and Karr 1994, Fore et al. 1996, Barbour et al. 1996)
% *Cricotopus*	Percent of midge larvae from the genus *Cricotopus*	X		**Increase, 5** (after Yoder and Rankin, 1995)
% Organic/Nutrient/DO Tolerant Taxa	Percent organic/nutrient/DO tolerant taxa	X		**Increase, 35** (after Yoder and Rankin, 1995)
% Toxics Tolerant Taxa	Percent toxic tolerant taxa.	X		**Increase, 35** (after Yoder and Rankin, 1995)

Total Number of Taxa (Taxa Richness)

Taxa richness, number of taxa present in a sample, or the variety of taxa, reflects community health and generally decreases with decreasing water quality or habitat suitability. Taxa richness on Hester-Dendy multiplate samplers (HD) increased between the Lockport and Brandon Pools (Appendix E Figure 1). Below the Brandon dam the number of taxa was highly variable. The distribution of taxa richness values between Lockport, Brandon and Upper and Lower Dresden Pools suggest similar taxa richness (Appendix E Figure 2). The increase in taxa richness from the Lockport to the Brandon Pool is likely the result of drift of organisms from the Upper Des Plaines River that enters the system in the Upper Brandon Pool.

Taxa richness in soft sediment samples, collected with a Ponar grab sample (PG), was quite low in Lockport and Brandon Pools (Appendix E Figure 3). This probably reflects the disturbance from barge traffic in the Pool. An increase in taxa richness downstream of the Brandon dam may reflect a more stable bottom substrate, not as prone to disturbance from barge traffic. Taxa richness in the Upper Dresden Pool was higher than the Lower Pool (Appendix E Figure 4). The trend suggests that the Upper Dresden Pool can meet similar taxa richness values found in the lower Pool.

EPT Taxa Richness

Number of EPT taxa in the sample, summarizes the taxa richness of pollution-sensitive species within the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). Generally EPT taxa richness will decrease with increasing perturbation. Perturbation is stress on the stream environment either in the form of physical disturbance or chemical toxicity. EPT taxa richness on HD samplers was low from the Lockport Forebay to the I-55 Bridge, except for station RM 282.8 which had eight EPT taxa (Appendix E Figure 5). Stations 277.6A and 277.6B, in the Lower Dresden Pool, had nine and seven EPT taxa, respectively. Most stations generally had less than 2 EPT taxa. EPT richness in Lower Dresden Pool was higher than Upper Dresden Pool, Brandon Pool, and Lockport Forebay, suggesting potential impairment in each of these river reaches, except for Station RM 282.8. Low numbers of samples in the Upper Dresden Pool as well as high variability in the data set precludes any definitive response, but the trend is for higher EPT taxa richness in the Lower Dresden Pool (Appendix E Figure 5).

It should be noted that variability in the data exists depending on where the HD samplers were placed. Table 5.5 provides an example of the data collected by IEPA at their station G-01, just downstream of the I-55 Bridge at RM 277.6 and 277.0. From the data we do see variability depending if the sampler was placed in the main channel or the channel border or tributary delta.

TABLE 5.5

Comparison of Sampling Results Depending on Sampler Location

River Mile	Major Habitat Type	EPT Taxa	Total Taxa	%EPT	MBI	ICI
277.6 A	Main Channel	9	24	26.7	6.1	22
277.6 B	Main Channel Boarder	7	20	34.4	5.3	24
277.0 A	Tributary Delta	3	14	53.2	5.2	16
277.0 B	Tributary Delta	2	8	68.0	5.2	12

Source: IEPA

EPT richness in soft sediment samples, collected with a Ponar grab sample (PG), was one or less (Appendix E Figure 7). There was no apparent trend in the data (Appendix E Figure 8).

Percent EPT Individuals

Percent composition of individuals in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) are compared to other invertebrates present in a sample. The abundance of the pollution-sensitive organisms will decrease with increasing perturbation. Percent EPT abundance on HD samplers was quite low in Lockport and Brandon Pools (Appendix E Figure 9). The abundance of these indicator organisms increased in a downstream direction. Lower and Upper Dresden Pools had similar abundance distributions (Appendix E Figure 10). EPT abundance in Lockport and Brandon Pools were lower than the Dresden Pool suggesting impairment.

Percent abundance of EPT taxa in soft sediment samples was low, indicating potential sediment issues (Appendix E Figures 11 and 12). Distinguishing a difference between each of the Pool areas was not possible using the Ponar grab data.

Total Number of Intolerant Benthic Taxa

Total number of taxa whose tolerance values are <6, based on Hilsenhoff's tolerance designations, are considered intolerant. The number of intolerant taxa will decrease with increasing perturbation. Number of intolerant taxa on HD samplers within the Brandon Pool averaged 9, while that in the Upper and Lower Dresden Pools averaged 9 and 8.5 taxa, respectively (Appendix E Figure 13). There was generally no difference in the number of intolerant taxa on HD samplers within each Pool (Appendix E Figure 14).

The number of intolerant taxa in the Lockport Forebay soft sediment sample was 2, while that in Brandon Pool averaged 1 (Appendix E Figure 15). The Upper Dresden Pool averaged 3 per station, and Lower Dresden Pool averaged 2. The number of intolerant taxa was low throughout the UAA assessment reach. A trend toward decreasing numbers of intolerant taxa in soft sediments in a downstream direction, except for Upper Dresden Pool, suggests potential impairment in the Lower Dresden Pool sediments (Appendix E Figure 16).

Percent Tolerant Individuals

Percent tolerant species is defined as the percent of individuals with tolerance values 6 based on Hilsenhoff's tolerance designations, compared to the total number of individuals present in the sample. Percent of tolerant individuals generally increase with increasing perturbation. The percent tolerant individuals on HD samplers was highest in the Lockport Forebay (83.4%) and in the Brandon Pool (average 63%). Brandon tailwater was also high at 72% (Appendix E Figure 17). Upper Dresden and Lower Dresden Pools had lower percent tolerant individuals at 49.5 and 37.8 percent, respectively. A reduction in tolerant taxa in a downstream direction suggests better physical or chemical conditions. The distribution of data points in each assessment area suggests that Upper and Lower Dresden Pools are similar, but Lockport and Brandon Pools are different from each other and the Dresden Pool (Appendix E Figure 18).

Percent tolerant individuals in soft sediment samples were nearly 100% in Lockport and Brandon Pools (Appendix E Figure 19). Although there was some decrease in the percent tolerant individuals in the Dresden Pool, most stations were above 80%. All assessment areas were quite similar (Appendix E Figure 20), and suggest potential impairment in soft sediments even in the Dresden Pool.

Family Chironomidae (Midge) Community Structure

The family Chironomidae represents the dominant group of benthic macroinvertebrates within the UAA reach. The chironomids as a group are generally considered more tolerant then mayflies, stoneflies, and caddisflies (EPT), although there are species within the family that are intolerant.

The number of taxa in the family Chironomidae generally will decrease as perturbation increases. The number of chironomid taxa on HD samplers was variable throughout the assessment reach (Appendix E Figure 21). Brandon Pool had the most species on average, while the Upper Dresden Pool had the least (Appendix E Figure 22). There were no clear-cut trends in the data set.

The number of chironomid taxa in ponar samples increased in Upper and Lower Dresden Pools (Appendix E Figure 23). There was good separation in distributions within each assessment area suggesting potential impairment in the Lockport and Brandon Pools (Appendix E Figure 24). Upper and Lower Dresden Pools were quite similar.

When chironomids begin to numerically dominate (one or two species become very abundant), to the exclusion of other taxa, this usually signifies impaired conditions. The percent Chironomidae on HD samplers in Lockport Forebay was 28.4%, that for the Brandon Pool was 26.8%, and for Upper and Lower Dresden Pool 35.5 and 40.5%, respectively. Lockport Forebay and Brandon Pool were generally the same (Appendix E Figures 25 and 26). Chironomidae abundance tended to increase in the Dresden Pool.

Percent Chironomidae abundance in ponar samples also tended to be low in Lockport and Brandon Pools, and increase in a downstream direction (Appendix E Figure 27). The highest numbers occurred in Lower Dresden Pool suggesting a potential increase in perturbation in soft sediments in a downstream direction (Appendix E Figure 28).

Within the family Chironomidae there are several subfamilies that may be of interest as indicator groups; these are the Chironominae, Tanypodinae, Podonominae, Diamesinae, Orthocladinae, and tribe Tanytarsini. The percent by major subfamily and tribe is the total number of individuals in each of the subfamilies to the total number of individuals in the Family Chironomidae. There were no Podonominae or Diamesinae in either HD or ponar samples; therefore, these two indicator groups were dropped from further consideration.

Chironominae were the dominant subfamily on HD samplers (Appendix E Figure 29). Chironominae distribution within each of the assessment reaches indicated no clear distinction between the Dresden Pool and that of Lockport or Brandon (Appendix E Figure 30).

There were few Chironominae in soft sediment samples, except for the Lower Dresden Pool (Appendix E Figure 31). The increase in the Lower Dresden Pool reflects what was seen in the Family Chironomidae metric, and suggests poor sediment quality. The distribution of Chironominae in Lockport and Brandon Pools was similar, while that of Upper Dresden was not similar to either the upper Pools or Lower Dresden Pool (Appendix E Figure 32).

The subfamily Orthocladinae associated with HD samples was highly variable throughout the UAA reach (Appendix E Figure 33). Generally the Orthocladinae will increase numerically as perturbation increases. The distribution of Orthocladinae within each assessment area did not segregate any of the assessment areas from each other (Appendix E Figure 34).

The Orthocladinae was not a dominate group within the family Chironomidae in soft sediment samples except for one station, RM 285.0, where it represented 78% of the Chironomidae abundance (Appendix E Figure 35). This metric was not assessed any further.

The subfamily Tanypodinae associated with HD samples generally represented a low proportion of the family Chironomidae. The use of this group as indicators of biological integrity is not well defined. Trends in the abundance data are highly variable and generally do not differentiate any of the assessment areas from each other (Appendix E Figures 36 and 37).

The Tanypodinae represent a high proportion of the Chironomidae community structure in soft sediment in both Lockport and Brandon Pools (Appendix E Figure 38). There is a reduction in the abundance of Tanypodinae in a downstream direction, especially in the Lower Dresden Pool (Appendix E Figure 39). The relationship between the abundance of Tanypods and sediment quality is not well defined.

The tribe Tanytarsini was not well represented on HD samplers (Appendix E Figure 40). There presence usually decreases as perturbation increases. The absence within the assessment reach may indicate poor water quality conditions.

Percent Composition by Major Group (other than Ephemeroptera, Plecoptera, Trichoptera, and Chironomidae)

Percent composition is the number of individuals within each major group compared to the total number of individuals collected in a sample. Major groups for consideration were aquatic worms (Oligochaeta), the family Hydropsychidae within the order Trichoptera, sow bugs (Isopoda), scuds (Amphipoda), dragonflies/damselflies (Odonata), and snails and bivalves (Phylum Mollusca).

Aquatic worms (Oligochaeta) generally flourish in conditions considered stressful for other groups. They are considered good indicators of organic enrichment. The percent aquatic worms on HD samplers was highest in Lockport Forebay (Appendix E Figure 41), and lowest in Lower Dresden Pool. A comparison of aquatic worm distributions between each assessment area suggests potential enrichment in the upper Pools and a decreasing trend in a downstream direction (Appendix E Figure 42). This metric appears to be a good indicator of possible organic enrichment within the UAA study reach.

Aquatic worm abundance was also high in the soft sediment samples, especially in Lockport and Brandon Pools (Appendix E Figure 43). The same trend seen in HD samples was also evident in soft sediment samples; high numbers of worms in the upper Pools and a subsequent reduction in a downstream direction (Appendix E Figure 44). The data suggests high organic enrichment, especially in the upper Pools.

Percent of Total Trichoptera as Hydropsychidae

This metric is the ratio of the number of individuals in the family Hydropsychidae to the total number of individuals in the order Trichoptera. Hydropsychid abundance will usually increase with an increase in perturbation and increase in fine particulate organic matter. No definitive pattern was evident in the data (Appendix E Figure 45). Comparison by area (Appendix E Figure 46) suggested a decrease in abundance in a downstream direction.

Percent Mollusca

The abundance of Mollusca organisms present is represented by snails and bivalves. Generally the percent Mollusca will decrease as perturbation increases (Kerns and Karr 1994, Fore et al. 1996, and Barbour et al 1996). Although Mollusca richness may decrease, several tolerant species may actually dominate the community structure (i.e. *Corbicula fluminea*). *Corbicula fluminea* represented a large percentage of the invertebrate community structure associated with HD samples at station RM288.3 (Appendix E Figure 47). Such a high percentage of *Corbicula fluminea* on HD samplers is unusual. Mollusca were more prevalent in the Brandon and upper Dresden Pools and decreased in a downstream direction (Appendix E Figure 48). This would suggest potential perturbation in a downstream direction, but the prominent tolerant bivalve, *Corbicula fluminea*, was the dominant organism. *Corbicula fluminea* decreased in the Lower Dresden Pool suggesting improved water quality.

The percent composition of Mollusca in soft sediment samples was inconsistent within the UAA study reach (Appendix E Figure 49). They were generally not very abundant. There was no segregation of assessment reaches (Appendix E Figure 50).

Percent Amphipoda

Amphipods will decrease with increasing perturbation. These organisms are associated with leaf detritus and slower currents. Although they may be found on HD samplers, this is not their preferred habitat. Distribution of amphipods within the study reach was highest in the Lockport Pool and lowest in the Dresden Pool (Appendix E Figure 51). The high numbers in Lockport may be due to the slower current. Likewise, the low numbers in Dresden Pool may be related to higher current velocities (Appendix E Figure 52).

Amphipods were only collected at three locations in soft sediment (Appendix E Figure 53). Further assessment of amphipod data was not warranted.

Percent Isopoda

Isopod abundance will increase with increasing perturbation. In the UAA study reach there were few isopods on HD samplers and none in soft sediment samples, therefore no further assessment was made on isopod data.

Percent Odonata

Dragonfly and Damselfly abundance will increase with increasing perturbation. The erratic distribution of odonats within the UAA study reach (Appendix E Figures 54 and 55) precluded further analysis.

Response Signature Metrics

Several metrics were suggested because of their use by Ohio EPA. These metrics were used only on HD data and represent part of a response signature by macroinvertebrate assemblages to disturbance. Three of these metrics were used to assess the response of the macroinvertebrate assemblage in the UAA study reach, although they are not used by OEPA in assessing channalized or impounded waters.

Percent Cricotopus sp.

This metric represents the percent of midge larvae of the genus *Cricotopus* sp. A macroinvertebrate assemblage with a Cricotopus sp. abundance greater than 5 percent may be considered impaired. Only the Lower Dresden Pool had an average Cricotopus sp. abundance greater than 5 percent (Appendix E Figures 56 and 57). This may be an artifact of a small sample size. There was no distinction between each of the assessment areas (Appendix E Figure 57).

Percent Organic/Nutrient/DO Tolerant Taxa

This metrices represents the percent abundance of organisms tolerant of organic loading, low dissolved oxygen and nutrient enrichment. Organic –tolerant taxa include Oligochaeta, *Glyptotendipes* (*G.*) sp. (not *G. barbipes*), *Chironomus* (*C.*) *decorus* group, *Chironomus* (*C.*) *riparius* group, *Dicrotendipes lucifer, Dicrotendipes neomodestus, Polypedilum* (*Tripodura*) *scalaenum* group, *Turbellaria*, *Physella* sp., *Simulium* sp.

Impairment is usually indicated when this metric exceeds 35 percent. Lockport and Brandon Pools exceeded the 35 percent limit Appendix E Figure 58). Only one station (RM 278.3) was above the limit in Upper Dresden Pool, and no stations were above this limit in Lower Dresden Pool. The distribution of values between each of the assessment areas suggests a reduction in impairment in a downstream direction (Appendix E Figure 59).

Percent Toxics Tolerant Taxa

This metrices represents the percent abundance of taxa designated by Ohio EPA as toxic tolerant taxa. Toxic-tolerant taxa include *Cricotopus* sp., *Dicrotendipes simpsoni*, *Glyptotendipes* (*G.*) *barbipes*, *Polypedilum* (*P.*) *fallax* group, *Polypedilum* (*P.*) *illinoense*, and *Nanocladius* (*N.*) *distinctus*.

Impairment is indicated when this metric value exceeds 35 percent. There was a trend in the data set towards an increase in percent toxics tolerant taxa in a downstream direction (Appendix E Figure 60). There were no values over 35 percent in Lockport, Brandon, or Upper Dresden Pools. There were several stations in the Lower Dresden Pool over the 35 percent limit. In general this metric did not do a good job of segregating the assessment areas (Appendix E Figure 61).

Conclusion of Individual Metrices Analysis

The use of individual community characteristics as tools for assessment of biological integrity were limited by the small sample size and lack of agreed upon reference conditions. In the absence of a reference condition, the Lower Dresden Pool was used as a comparison point only because it is currently classified as "General Use". Whether it is actually meeting that use is still a point of discussion. The following metrics segregated the assessment areas, or indicated no difference between areas when compared to Lower Dresden Pool, percent EPT taxa; number of intolerant taxa; percent tolerant taxa; percent Oligochaeta, percent Organic/Nutrient/DO tolerant taxa, and percent toxics tolerant taxa for HD sampling (water column); and taxa richness. The number of intolerant taxa, number of Chironomidae taxa, percent Chironomidae, percent Chironominae, percent Tanypodinae, and percent Oligochaeta appeared to be good metrics for assessing biological integrity in soft sediments. Some of these metrics indicated restricted community structure in the Lockport and Brandon Pools. The richness measurements suggest greater macroinvertebrate diversity in the Upper and Lower Dresden Pools.

In-situ samples of benthic macroinvertebrate collected through Ponar dredge sampling (PG) indicate that habitat is very limited in the study reaches. Sediments are frequently disturbed by barge traffic and the system is limited in riparian habitat and woody debris. The greatest lack of habitat exists in the Brandon Pool where the stream edge is channelized and lined with concrete retaining walls. In both the Brandon and Dresden Pools the water is impounded, reducing stream velocity and creating a deep water habitat that is not optimum for a diverse benthic macroinvertebrate community. Greater taxa richness, % EPT abundance, and percent tolerant organisms collected on artificial substrates (Hester-Dendy samplers) indicate that water quality could support a more diverse benthic community if adequate habitat was available.

Biological Indexes

Biological indexes are generally a composite of several metrics (community characteristics) that represent a biological condition (impaired, nonimpaired). The State of Illinois has several such indexes that use different biological components to assess biological integrity of its water resources. One such index, the Macroinvertebrate Biotic Index (MBI), uses benthic macroinvertebrates as an indicator component, and is based on only one community characteristic--tolerance. Another index that has been developed from a large data base and applicable to Illinois waters is Ohio's Invertebrate Community Index (ICI).

Macroinvertebrate Biotic Index (MBI)

Illinois' Macroinvertebrate Biotic Index (MBI) is a modification of the tolerance index developed by Hilsenhoff (1982) and was developed by the Illinois Environmental Protection Agency (IEPA, 1994) to provide a rapid stream quality assessment. The MBI is a single metric index that reflects the range of tolerances in a benthic community structure. Macroinvertebrate taxa known to occur in Illinois are assigned a pollution rating (tolerance value) based on references and field studies. The MBI is an average of the tolerance ratings weighted by organism abundance.

Comparison to Illinois General Use Criteria in 305b Report

The use of benthic macroinvertebrates as a tool to assess stream health is well documented in the literature. The essence of this tool is the comparison of known community traits to standards or reference criteria that reflect a desired water quality condition (best attainable condition). The outcome of this comparison is to predict whether the aquatic life use is being met, using existing biota as indicators, and thus the health of the water resource under investigation. This differs from water quality standards as stated in Illinois' Title 35: Environmental Protection, Part 303, Subpart B: Nonspecific Water Use Designations, which base use attainment on water quality parameters and narrative standards.

Section 305(b) of the federal Clean Water Act (CWA) of 1972, indicates that each state is required to prepare and submit to the U.S. Congress and the U.S. EPA a biennial report which includes:

An assessment of the water quality for surface and groundwater resources;
An analysis of the extent to which such waters provide for the protection and propagation of shellfish, fish, and wildlife as well as allow for recreational activities;
An estimate of the environmental impacts, costs and benefits, and time frame to achieve the requirements of the CWA; and
A description of the nature and extent of nonpoint source pollution and recommendations to address this pollution.

To this end, Illinois develops a 305(b) report to assess overall water quality in its streams, rivers, and lakes. In the 2000 305(b) assessment documentation, a process was developed to assess the health of surface water resources using biological, physical, as well as chemical parameters to assess aquatic life use attainment. One of the biological criteria used is the MBI. The aquatic life use for General Use Waters is fully supported when an MBI less than 5.9 is attained. The use is considered partially supported with an MBI of 6.0 to 8.9, and is considered not supported with an MBI greater than 9.0 (IEPA 1999).

Invertebrate Community Index (ICI)

The Ohio Invertebrate Community Index (ICI) is a multimetric index that uses several benthic macroinvertebrate community characteristics to assess attainment of aquatic life use based on biological performance (OEPA 1987, Yoder and Rankin 1995). This index differs significantly from the MBI in that the MBI is a single value index that reflects only one biological trait (tolerance). The strength of the multimetric process is the use of many biological traits that reflect potential changes in community structure or function in relation to impairment.

The ICI is a modification of the Index of Biotic Integrity (IBI) for fish developed by Karr (1981). The ICI consists of 10 structural community metrics, each with four scoring categories of 6, 4, 2, and 0 points (Table 5.6). The point system evaluates a sample against a database of 247 relatively undisturbed reference sites throughout Ohio. Six points will be scored if a given metric has a value comparable to those of exceptional stream communities, 4 points for those metric values characteristic of more typical good communities, 2 points for metric values slightly deviating from the expected range of good values, and 0 points for metric values strongly deviating from the expected range of good values. The summation of the individual metric scores (determined by the relevant attributes of an invertebrate sample with some consideration given to stream drainage area) results in the ICI value. Metrics 1 through 9 are all generated from the artificial substrate sample data, while Metric 10 is based solely on the qualitative sample data from natural substrates. More discussion of the derivation of the ICI including descriptions of each metric and the data plots and other information used to score each metric can be found in Ohio EPA (1987).

TABLE 5.6
Metrics Used in the Calculation of the
Ohio Invertebrate Community Index (ICI)

Metric	Scoring[1]
1. Total number of taxa	0 2 4 6
2. Total number of mayfly taxa	0 2 4 6
3. Total number of caddisfly taxa	0 2 4 6
4. Total number diperan taxa	0 2 4 6
5. Percent mayflies	0 2 4 6
6. Percent caddisflies	0 2 4 6
7. Percent tribe tanytarsini midges	0 2 4 6
8. Percent other diperans and non-insects	0 2 4 6
9. Percent tolerant organisms	0 2 4 6
10. Total number of qualitative Ephemeroptera, Plecoptera, and Tricoptera (EPT) taxa	0 2 4 6

[1] See Ohio, 1987

Use of MBI and ICI to Assess Illinois General Use Classification

Macroinvertebrate data collected from the Des Plaines River UAA Reach were converted to MBI's and compared to aquatic life support assessment criteria (IEPA 1999). Hester-Dendy data indicated that all sample locations in the Lower Dresden Pool, except one, met the general use classification (Appendix E Figures 62 and 63). About half of the Upper Dresden Pool stations were fully meeting the general use classification. Several locations were partially supporting and no locations were considered non-supporting. Of the stations in the Lockport and Brandon Pools, all were considered partially supporting.

Macroinvertebrate data from soft sediment samples were also converted to MBI's and compared to aquatic life support assessment criteria (IEPA 1999). Most sample locations were considered non-supporting (Appendix E Figure 64). Comparison between UAA areas (Appendix E Figure 65) indicated that all areas were considered non-supporting. This has already been suggested based on individual metrics.

The Hester-Dendy data was converted to ICI values. A comparison of the data to the Ohio criteria for Warmwater Habitat (WWH) (ICI 30 to 36 depending on state ecoregion) and Modified Warmwater Habitat (MWH) Channel Modified (ICI 22) aquatic life use categories was made. In all cases ICI values indicated that neither criteria was supported (Appendix E Figures 66 and 67). It should be noted that Ohio does not have ICI criteria for impounded waters due to a concern for a lack of adequate stream velocity. According to Ohio EPA (1987) the current should be no less than 0.3 ft/sec in order to properly use the ICI. In the Lower Des Plaines River study area average velocities are 0.75 ft/sec and 0.65 ft/sec in the Brandon and Dresden navigation pools respectively, allowing comparison to Ohio's Channel Modified criteria.

Summary

Benthic macroinvertebrates were investigated as indicators of aquatic life use in this UAA. The use of benthic macroinvertebrates to assess biological integrity is well documented, therefore, warranted in this UAA.

Macroinvertebrate metrics collected using Hester-Dendy (HD) artificial samplers suggested a general trend of improved water quality from upstream to downstream. Based on artificial substrates and use of the Illinois single matrix MBI, the Upper Dresden Pool appears to provide water quality sufficient to support a General Use Classification. The use of the Ohio multi-metric ICI indicates that the Upper Dresden Pool is not meeting its potential use as impounded water. The macroinvertebrate community in the Brandon Pool does not support a General Use Classification and both the Illinois MBI and Ohio ICI indicate a degraded macroinvertebrate community.

Samples of benthic macroinvertebrate collected through Ponar dredge-sampling (PG) show a much more degraded condition as compared to samples collected on artificial substrates. Illinois MBI values for all of the study reaches indicate a benthic community that does not meet the General Use Classification. Benthic habitat in the entire study area has limited epifaunal substrate suitable for invertebrates, including woody debris, cobbles, stable substrate, and under cut banks. In both the Brandon and Dresden Pools the water is impounded, reducing stream velocity and creating deep-water habitat that is not optimum for a diverse benthic macroinvertebrate community. Impoundments are typically characterized by fine-grained bed material (Petts, 1984). The heterogeneity of the channel-bed sediments, in terms of size, is of critical importance in providing microhabitats, which can support abundant and diverse fauna (Hynes, 1970). Sediments in the federal navigational channel are frequently disturbed by barge traffic (Butts and Shackleford, 1992; Bhowmik et al., 1989). Disturbance of the sediments impedes colonization of benthic organisms. The greatest lack of habitat for macroinvertebrate exists in the Brandon Pool where the stream edge is channelized and lined with concrete retaining walls.

These conclusions are based on a limited set of data and should be viewed judiciously. In addition, there was no agreed upon reference condition to which the data could be compared. However, quantitative comparisons were limited to existing statewide index values, which were developed for small streams and not large river impoundments. The results of the macroinvertebrate sampling were heavily influenced by lack of habitat and barge traffic. Results of the macroinvertebrate analysis need to be viewed as only one component of the "weight of evidence" needed to draw conclusions about the current biological use of the Lower Des Plaines River.

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CHAPTER 6

EVALUATION OF EXISTING AND POTENTIAL FISHERY COMMUNITY

Introduction

Analysis of fish community structure has long been recognized as tool for assessing the quality of an aquatic community. Attributes of fish assemblages are useful for assessing stream quality because fish represent the upper level of the aquatic food chain and thus reflect conditions in lower trophic levels (e.g. primary producers and consumers) (Karr 1981, Karr et al. 1986, Bertrand et al., 1996). Fish more than any other biological indicator display the ability to integrate stress from both chemical and habitat perturbations associated with both point and nonpoint source pollution. Other factors that make fish useful in qualitative assessments include their ease of identification, public recognition of their importance, the availability of information stress and acute toxicity effects, and extensive life history information (Karr et al.1986). Fish data can be linked to attainment of the "fishable-swimable" goals outlined in Section 101(a)(2) of the federal Clean Water Act.

Dr. Philip Smith noted that mere presence of fish provides little information about the condition of a stream, but "knowledge of the assemblage of species and their numerical relationships" provides "an excellent biological picture of the water course and its well being" (Smith, 1971). Karr et al. (1981) developed the Index of Biotic Integrity (IBI) to assess the biological integrity of low gradient warmwater streams affected by agriculture in the Midwestern U. S. The IBI was revised in 1986 (Karr et al. 1986). The IBI incorporates 12 aggregations of community information termed "metrics". The metrics fall into three broad categories: species richness and composition, trophic composition and fish abundance (Yoder and Rankin, 1995). Some metrics respond positively (i.e., their raw value increases) to environmental quality and are termed positive metrics. Other metrics respond positively to increase degradation (i.e., their raw value decreases) and are termed negative metrics. Some metrics respond across the entire range of environmental quality where as others respond more strongly to a portion of that range (Karr et al. 1986). While no single metric can consistently function across all types of impacts, the aggregation of metrics combined in the IBI provides sufficient redundancy to provide a consistent and sensitive measurement of biological integrity (Angermire and Karr et al. 1986). The IBI relies on multiple parameters; an essential attribute when the system being evaluated is complex (Karr et al. 1986). While the IBI incorporates elements of professional judgment, it also provides the basis for establishing quantitative criteria for determining what constitutes exceptional, good, fair, poor, and very poor conditions (Yoder and Rankin, 1995).

Description of Indices of Biotic Integrity

To evaluate stream quality at the community level, Karr (1981) proposed and revised (Karr et al. 1986) the Index of Biological Integrity (IBI). The IBI is comprised of 12 metrics to define community structure. The index accounts for changes in community richness and allows for comparison of fish community composition with maximum known values of similar-sized streams. The applicability of the IBI concept has been demonstrated in a wide variety of stream types (Miller et al., 1988). As recommended by Karr et al. (1986), IBI metrics require adjustment for the region to which the index is applied.

Illinois IBI

The State of Illinois uses the IBI generally unaltered from the original index developed by Karr et al. (1986). The index is outlined in Table 6.A.

Table 6.A
Illinois Index of Biological Integrity (IBI)

Category	Metrics	Scoring Criteria		
		5	3	1
Species Richness and Composition	1. Total number of fish species	Expectations for metrics 1-5 vary with stream size and region. Tables of appropriate values for seven IBI regions in Illinois are summarized in Appendix A of Bertrand et al. (1996)		
	2. Number and identity of darter species			
	3. Number and identity of sunfish species			
	4. Number and identity of suckers species			
	5. Number and identity of intolerant species			
	6. Proportion of individuals as green sunfish	<5%	>5-20%	>20%
Trophic Composition	7. Proportion of individuals as omnivores	<20%	>20-45%	>45%
	8. Proportion of individuals as insectivorous cyprinids	>45%	<45-20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	<5-1%	<1%
Fish Abundance and Condition	10. Number of individuals in sample	Expectations for metrics 1-5 vary with stream size and region. Tables of appropriate values for seven IBI regions in Illinois are summarized in Appendix A of Bertrand et al. (1996)		
	11. Proportion of individuals as hybrids	0%	>0-1%	>1%
	12. Proportion of individuals with disease, tumors, fin damage, skeletal anomalies	0-2%	>2-5%	>5%

Source: Bertrand et al. (1996)

It is recognized that stream size is an important factor when refining the IBI to a geographical region. The Illinois IBI has generally been well calibrated to small wadable streams. However, a large stream index for use on waterways such as the Lower Des Plaines River has not been calibrated for Illinois at this time.

Ohio IBI

The State of Ohio Environmental Protection Agency (OEPA), similar to Illinois, has developed a series of IBI values based on regions of similar characteristics. While the Illinois IBI calibration focused primarily on smaller wadable streams, the OEPA added a "Boatable" stream category to their IBI system. Ohio developed three different modified IBI's, all based on the basic ecological structure and content of Karr's original IBI. The indexes are for headwater streams IBI (defined as stream locations with a drainage area <20 square miles), a wading site IBI applicable to streams >20 square miles sampled with wading methods, and a boatable site IBI for locations that need to be sampled with boat methods. Boatable sites include large rivers similar to the Lower Des Plaines River. The IBI divisions were made based on inherent difference in faunal associations (e.g., headwaters vs. wading sites) and sampling gear bias considerations (e.g., wading vs. boatable sites) (Yoder and Rankin, 1995). Table 6.B summarizes the modifications made to Karr's original IBI for wading and boatable sites.

After an analysis by the Lower Des Plaines River Use Attainability Analysis Biological Subcommittee, it was decided that the Ohio Boatable IBI was the most appropriate index for evaluation of the Lower Des Plaines River. The Ohio Boatable IBI had been calibrated for use on large rivers that had been sampled using the methods applied to past studies on the Lower Des Plaines River.

Trends in Fisheries Data

Data Collection and Analysis Methods

The fish community of the Lower Des Plaines River was sampled by scientists from EA Engineering, Science and Technology (EA), on behalf of Commonwealth Edison Company or Midwest Generation EME, LLC. The EA sampling was conducted using the methods prescribed by the Ohio IBI methodology. While the Lower Des Plaines River has been sampled in the past by the Illinois Department of Natural Resources (IDNR), the purpose of the sampling was to determine abundance of sport fish species and was not designed to assess community structure. Therefore only data collected by EA will be used for the following analysis.

Table 6.B

Modification of Index of Biotic Integrity (IBI) metrics used by OEPA to Evaluate Headwater, Wading and Boatable Sites (the original IBI metrics of Karr 1981 are given first with substitute metrics following)

IBI Metric	Headwater Sites[1]	Wading Sites[2]	Boatable Site[3]
1. Number of native fish species[4]	X	X	X
2. Number of darter species		X	
Number of darter and sculpin species	X		
% round-bodied suckers [5]			X
3. Number of sunfish species[6]		X	X
Number of headwater species[7]	X		
4. Number of suckers species		X	X
Number of Minnow species	X		
5. Number and identity of intolerant species		X	X
Number of sensitive species[8]	X		
6. % green sunfish			
% tolerant species	X	X	X
7. % omnivores	X	X	X
8. % insectivorous cyprinids			
% insectivores	X	X	X
9. %piscivores (top carnivores)		X	X
% pioneering species[9]	X		
10. Number of individuals			
Number of individuals (minus tolerant)[10]	X	X	X
11. % hybrids			
% simple lithophils		X	X
Number of simple lithophils	X		
12. % diseased individuals % DELT anomalies[11]	x	x	x

Source : Yoder and Rankin, 1995

[1] applies to sites with drainage areas<20 square miles.
[2] sampled with wading electrofishing methods.
[3] sampled with boat electrofishing methods.
[4] excludes all exotic and introduced species.
[5] includes all species of the genera *Moxostoma, Hypente;ium, Minytrema* and *Ericymba*, and excludes *Catastomus commersoni.*
[6] includes only Lepomis species.
[7] species designated as permanent residents of headwater streams.
[8] includes species designated as intolerant and moderately intolerant (Ohio EPA 1987).
[9] species designated as frequent and predominant inhabitants of temporal habitat in headwaters streams.
[10] excludes all species designated as tolerant, hybrids, and non-native species.
[11] includes only individuals with deformities, eroded fins or barbells, lesions, and tumors.

Sampling was conducted at 20 locations using a boat-mounted electrofishing system and a variety of habitat types, including encompassing main channel, main channel border, dam tailwater, and tributary mouth habitats.

The study area was divided into four reaches (from downstream to upstream): **(1)** ***lower Dresden Pool***, General Use waters of the Des Plaines River from the confluence with the Kankakee River up to the I-55 Bridge (4 stations); **(2)** ***upper Dresden Pool***, Secondary Contact waters of the Des Plaines River from the I-55 Bridge upstream to the Brandon Road Lock and Dam (5 stations); **(3)** ***Brandon Pool***, Secondary Contact waters of the Chicago Sanitary Ship Canal (CSSC) and Des Plaines River between the Lockport and Brandon Dams (7 stations), and **(4)** ***lower Lockport Pool***, Secondary Contact waters of the CSSC upstream of Lockport Dam (4 stations). Data from 1999, 2000, and 2001 was used. Not all stations were sampled in 1999. Dates of sampling varied among years. Details of the sampling methods, data collection, and sampling station descriptions are presented in 2000 Upper Illinois Waterway Fisheries Investigation, RM 274.4-296.4 (EA, 2001).

Sampling methodology and analysis procedures followed those outlined by the state of Ohio Environmental Protection Agency Ecological Assessment Section document "Biological Criteria for the Protection of Aquatic Life: Volume II: Users Manual for Biological and Field Assessment of Ohio Surface Waters, Updated January 1, 1988." (Ohio EPA, 1989). Fish classifications were assigned based upon those used by the Illinois EPA (Bertrand, et al. 1993) and Ohio Boatable IBI scores were adjusted for low catch rates when necessary. Electronic copies of the fisheries data and calculations of the Ohio IBI metrics were provided by EA. Data summaries for all stations and sampling dates for 1999-2001 are presented in Appendices 6.1-6.3. For analysis, data were summarized both by river mile along the study area and by sampling reach. Box plots and box and whisker plots provide a convenient way to visualize the spatial relationships among sampling stations (Figure 6.1) and are used to present the data.

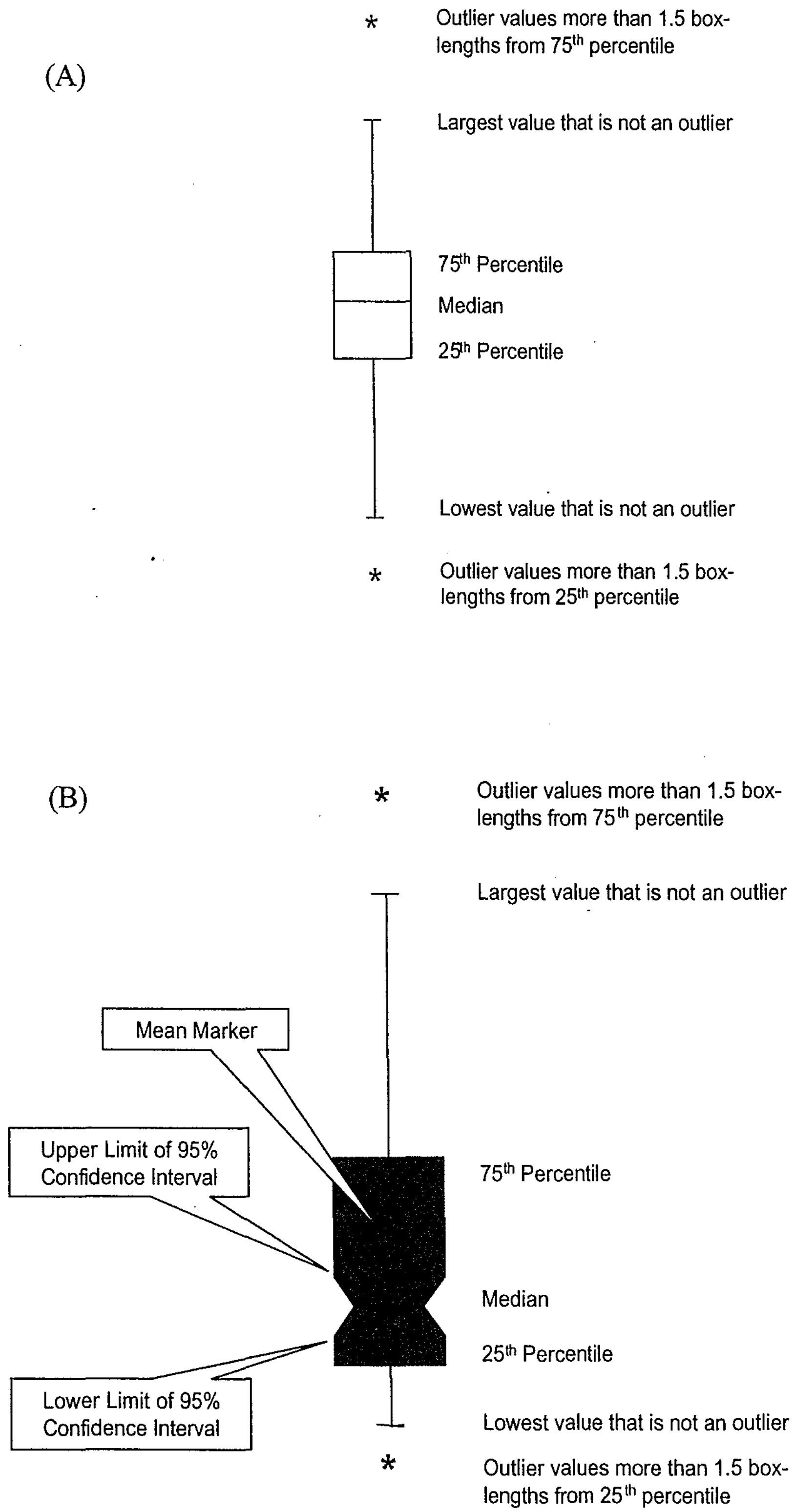


Figure 6.1 Diagrams illustrating the components of (a) a Box Plot and (b) a Box and Whisker Plot

Spatial and Temporal Trends in IBI

Ohio IBI values were calculated for fish samples collected for stations in the Lower Des Plaines River from 1999-2001 and pooled for all sampling dates. Summary charts of the data are presented in Figures 6.2-6.4.

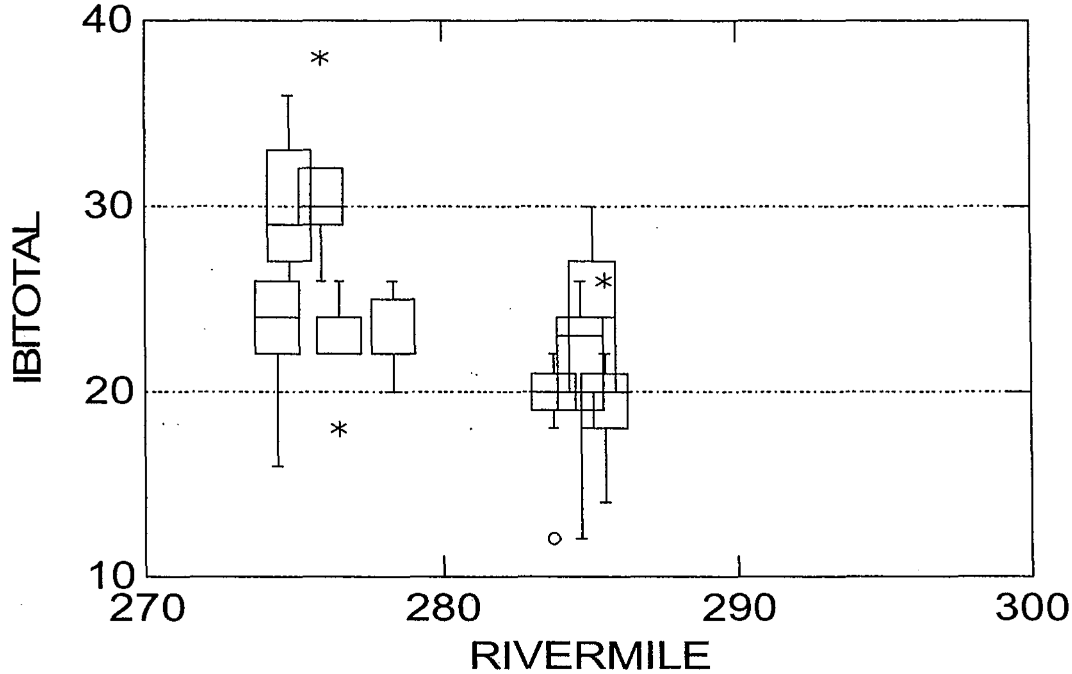
When plotted by river mile, there is a consistent trend showing a decrease in IBI from downstream to upstream in all three years (Figures 6.2a, 6.3a, and 6.4a). This corresponds to a general pattern of declining IBI among reaches (Figures 6.2b, 6.3b, and 6.4b), with Lower Dresden higher than Upper Dresden, and both Dresden Reaches higher than both Brandon and Lockport Pools.

Because the charts in Figures 6.2-6.4 show data pooled for each sampling station for multiple sampling dates, much of the variation observed may be attributable to seasonal "time-of-year" effects, independent of differences due to location in the river. In order to test for differences among Reaches, a split-plot analysis of variance (ANOVA) was used to partition variance in IBI scores among three factors; (1) Reach, (2) Year, and (3) Month nested within Year. The ANOVA allows for the differences in IBI among reaches to be compared statistically, separate from any underlying seasonal effects. The results of the ANOVA analysis are presented in Table 6.1 and show a very significant effect of all three factors on IBI ($p < 0.001$). This means, in essence, that there were differences in IBI among years and months within years which could be due to numerous effects such as natural climatic variation, temperature effects, fish migration patterns, etc. However, this also means that even after considering yearly and seasonal variation, there are still consistent and significant differences among the four reaches.

A Post-hoc Multiple Comparisons Analysis was conducted to test for pair-wise differences among Reaches. This test applies a multiple comparison procedure to determine which means are significantly different from which others. Results are shown in Table 6.2. The bottom half of the output shows the estimated difference between each pair of means. An asterisk has been placed next to 5 pairs, indicating that these pairs show statistically significant differences at the 95.0% confidence level. In the upper half of Table 6.2, three homogenous groups are identified using columns of X's. Within each column, the levels containing X's form a group of means within which there are no statistically significant differences. A fisher's least significant difference (LSD) procedure was used to discriminate among means, with a p-Value of 0.05 (i.e. a 5.0% risk of calling each pair of means significantly different when there is no actual difference).

The mean IBI values $\pm$ 95% confidence intervals for each Reach (based upon ANOVA with all sampling dates included) are shown in Figure 6.5. Mean IBI values are 23.79 for Lower Dresden, 20.51 for Upper Dresden, 17.40 for Brandon Pool, and 16.45 for Lockport. The analysis shows that Lockport and Brandon Pools are not statistically different from each other, whereas both Lockport and Brandon have significantly lower IBI than both Upper Dresden and Lower Dresden. Furthermore, Upper Dresden has a significantly lower IBI than Lower Dresden.

(A)



(B)

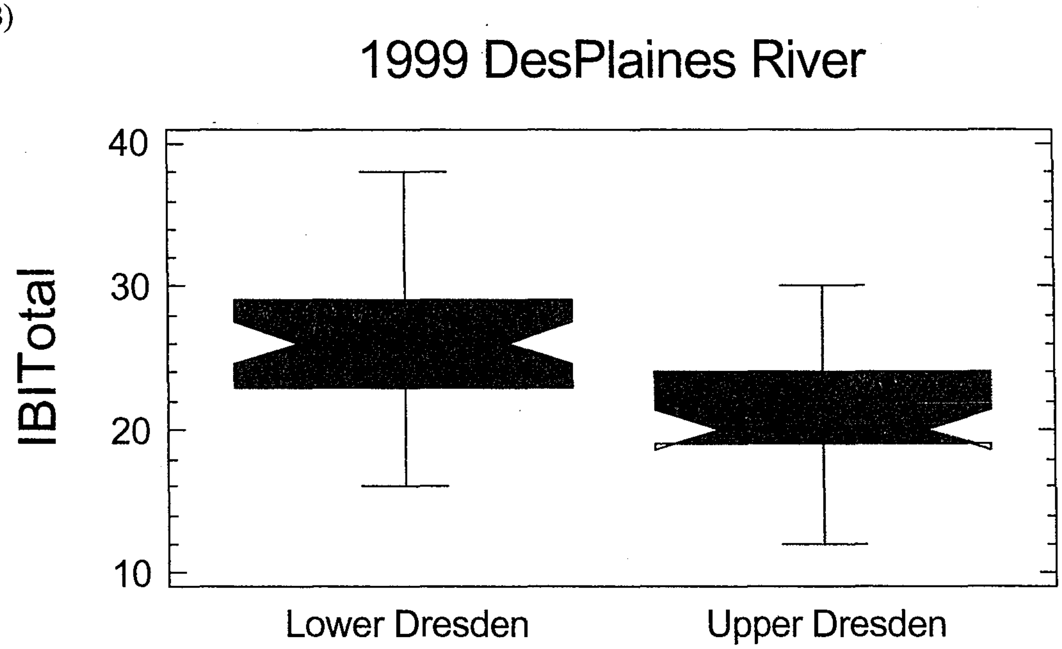
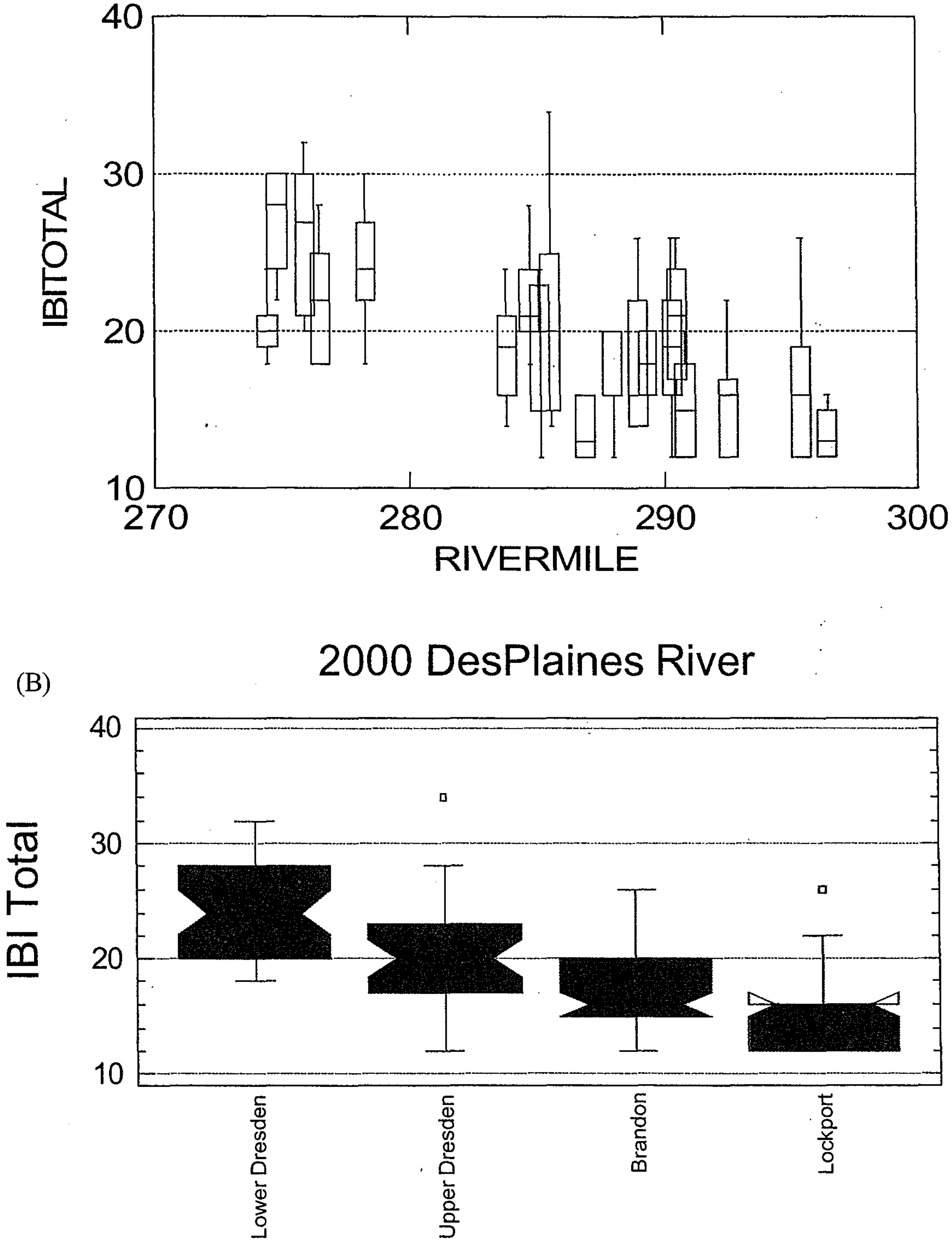


Figure 6.2 Ohio IBI calculated for sampling stations in the Lower Des Plaines River for 1999. (a) Sampling stations pooled by River Mile of the station, (b) Sampling stations pooled by Reach.

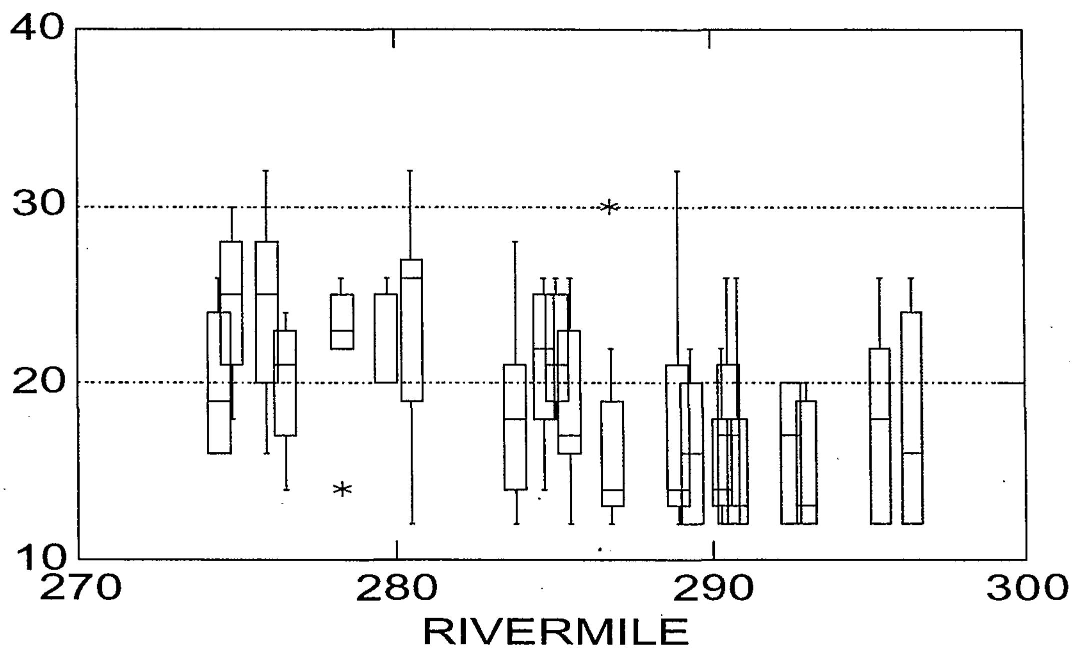
(A)



(B)

Figure 6.3 Ohio IBI calculated for sampling stations in the Lower Des Plaines River for 2000. (a) Sampling stations pooled by River Mile of the station, (b) Sampling stations pooled by Reach.

(A)



(B)

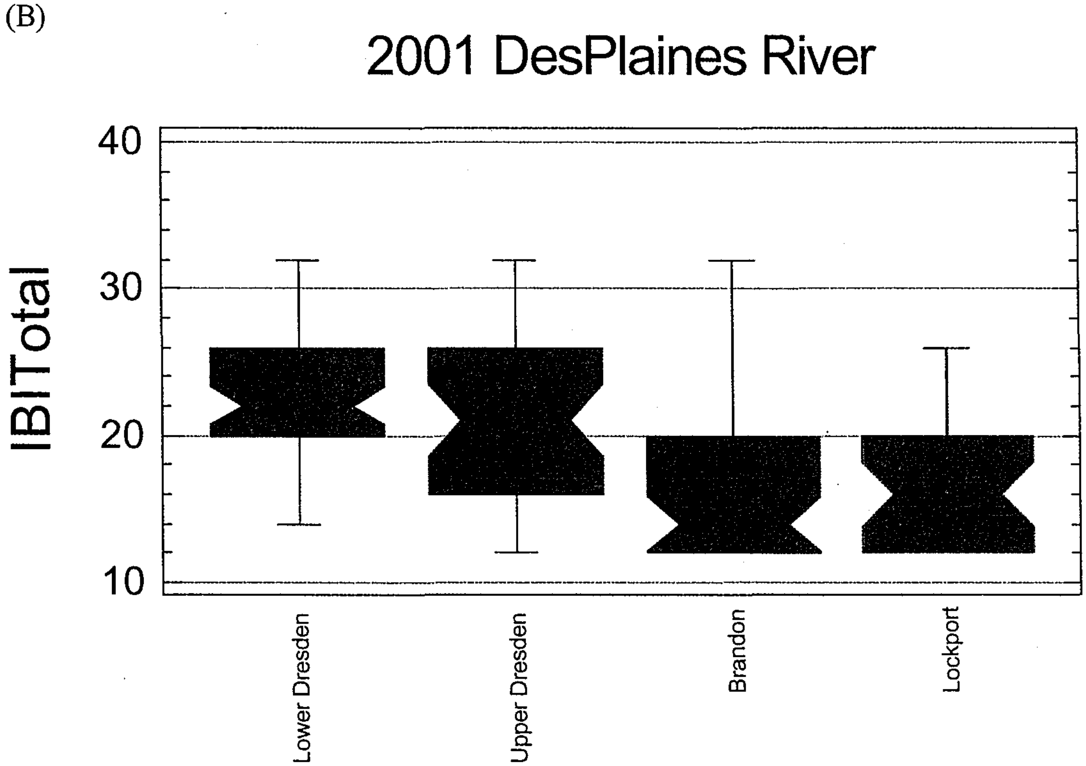


Figure 6.4 Ohio IBI calculated for sampling stations in the Lower Des Plaines River for 2001. (a) Sampling stations pooled by River Mile of the station, (b) Sampling stations pooled by Reach.

Table 6.1 : Analysis of Variance for IBI for Lower Des Plaines River

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
Model	5681.21	18	315.623	17.94	0.0000
Residual	6686.75	380	17.5967		
Total (Corr.)	12368.0	398			

Type III Sums of Squares

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
Reach	2911.65	3	970.549	55.16	0.0000
Year	296.217	2	148.109	8.42	0.0003
Month(Year)	1484.47	13	114.19	6.49	0.0000
Residual	6686.75	380	17.5967		
Total (corrected)	12368.0	398			

Table 6.2: Multiple Comparisons for IBI Total by Reach following ANOVA

Method: 95.0 percent LSD

SEGMENT$	Count	LS Mean	LS Sigma	Homogeneous Groups
Lockport	64	16.4461	0.566012	X
Brandon	104	17.3937	0.463896	X
Upper Dresd	104	20.513	0.419073	X
Lower Dresd	127	23.7898	0.380295	X

Contrast	Difference	+/- Limits
Lower Dresd - Upper Dresd	*3.27675	1.09089
Lower Dresd - Brandon	*6.39608	1.15394
Lower Dresd - Lockport	*7.34365	1.31774
Upper Dresd - Brandon	*3.11933	1.20218
Upper Dresd - Lockport	*4.0669	1.35994
Brandon - Lockport	0.947572	1.31086

* denotes a statistically significant difference.

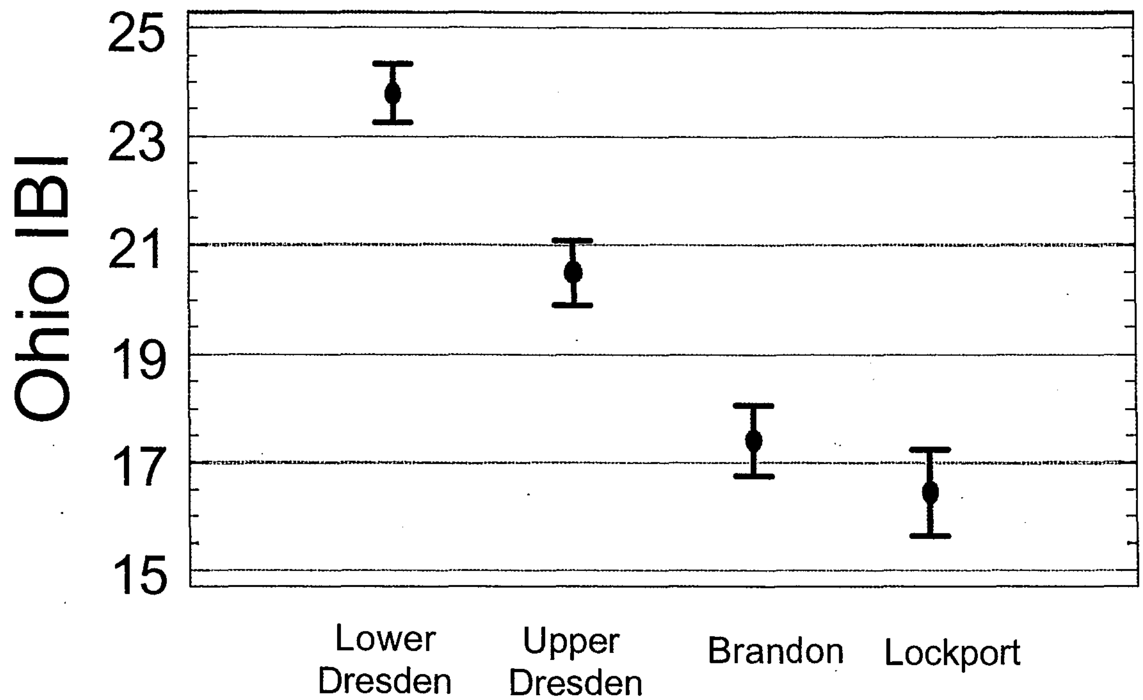


Figure 6.5 **Mean IBI ± 95% confidence interval for Ohio IBI Scores calculated for four reaches in Des Plaines River and Chicago Sanitary Ship Canal. Means are least-square means following ANOVA for effect Reach, Year, and Month nested within Year.**

Analysis of Individual Metrics Contributing to IBI Scores for the Lower Des Plaines River

Although the multimetric Index of Biotic Integrity is useful in providing a concise summary of complex ecological information inherent in aquatic communities, careful examination of the contributing metrics of the IBI can provide additional insight into potential stressors operating within the system (Rankin and Yoder, 1995).

Box Plots of the 12 metrics used to calculate the Ohio IBI, pooled by Reach for all sampling dates from 2000 and 2001, are presented in Figure 6.6. Several of the species composition metrics exhibit clear spatial patterns. The numbers of native species (NATIVE) and sunfish species (SUNFISH) show consistent and significant declines moving upstream from Lower Dresden in each successive reach (Table 6.5A and 6.5B, Multiple Comparison ANOVA by Year and Reach, $p<0.05$). Since sunfishes are largely non-migratory, this pattern suggests a progressively graded stressor or suite of stressors in the system. By contrast, the numbers of sucker species (SUCKER), which tend to be more mobile and migratory by nature, are similar in Upper and Lower Dresden but drop drastically above the Brandon Dam ($p < 0.05$). This suggests either that the Brandon Dam may be a barrier to fish movement in the system, or that the change in habitat upstream from Brandon Dam makes the system unsuitable for sucker species.

Intolerant species (INTOLERANT) were very rare or absent in all samples, and the abundance of individuals, not including tolerant species (NONTOLCPE), was significantly higher in Lower Dresden than the other reaches ($p < 0.05$). This suggests that chronic stresses such as poor habitat, thermal and/or oxygen stress may be impacting fish communities. This is further supported by the spatial trend in percentage of fish that are tolerant species (TOLERANTPCT), with Upper Dresden and Brandon Pool higher compared to Lower Dresden. A Multiple Comparisons ANOVA test for percent tolerant individuals (Table 6.6A) shows that Lower Dresden had significantly lower percentage of tolerant fish compared to the three other reaches.

The percentage of all individuals that were either round bodied suckers (RBSKRSPCT) or top carnivores (TOPCARNPCT) was higher in Upper and Lower Dresden compared to Brandon and Lockport Pools, but the higher variance in these metrics (due in part to the disproportionate sensitivity of small percentages to the effects of low fish numbers for some sampling stations) makes these trends difficult to interpret. Likewise, the percent of individuals that are omnivores (OMNIPCT) was higher in Brandon and Lockport compared to Upper and Lower Dresden.

The percentage of individuals that were insectivores (INSECTPCT), or simple lithophilic spawners (LITHOPHPCT), or that exhibited DELT anomalies (DELTPCT) was highly variable within and among reaches and did not exhibit clear spatial trends. However, a Multiple Comparisons ANOVA test for DELT anomalies, including difference among years (Table 6.6B), shows that Lower Dresden had significantly lower DELT percentages than both Upper Dresden and Brandon Pool.

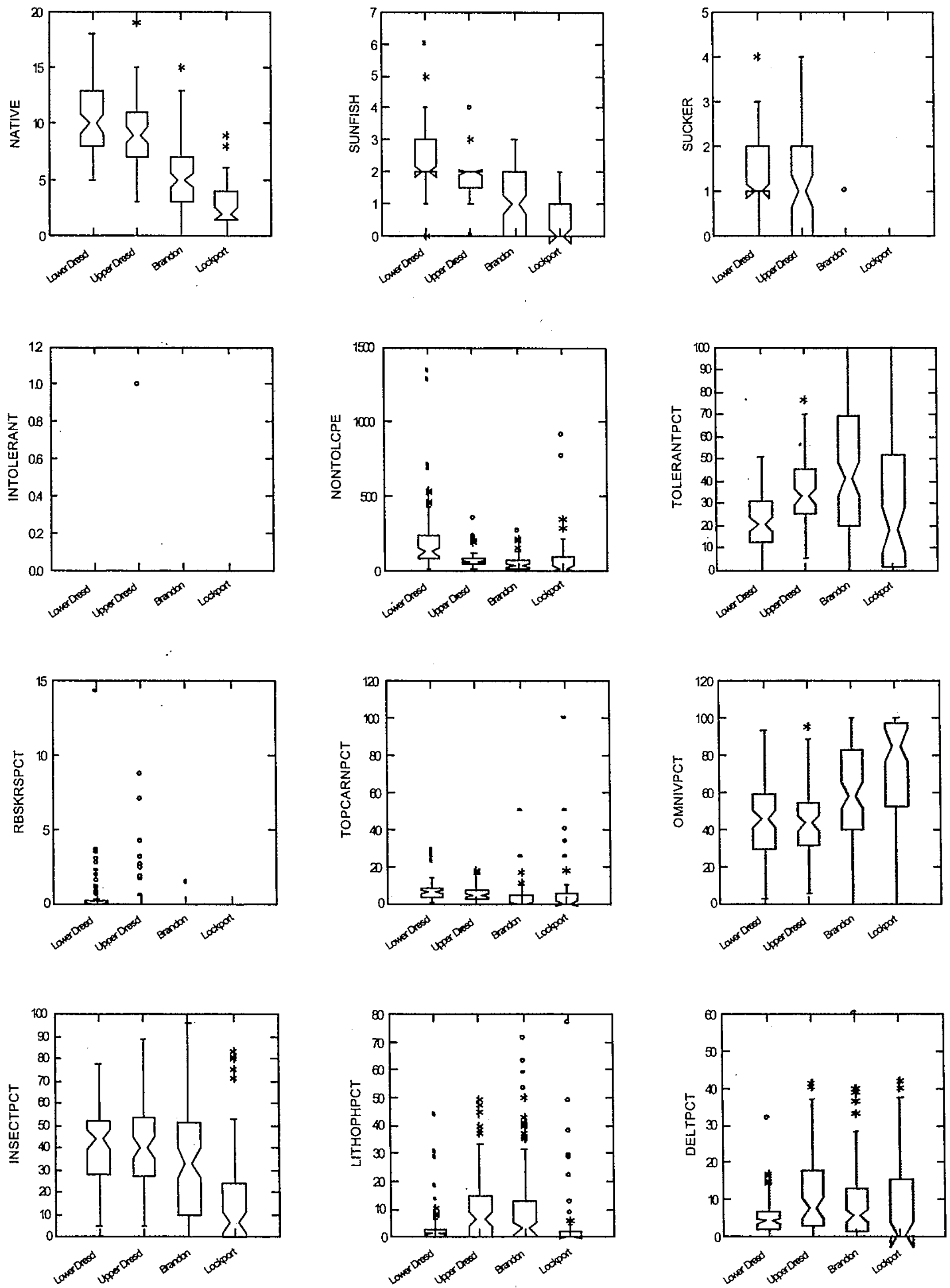


Figure 6.6 Examination of Contributing Metrics used to calculate Ohio Boatable IBI. Data are pooled for all sampling dates from 2000 and 2001.

Table 6.5 Multiple Comparisons Test for Numbers of Species by Reach in the Lower Des Plaines River that are (A) Native Species and (B) Sunfish Species.

(A) Multiple Range Tests for NATIVE by Reach$

Method: 95.0 percent LSD

SEGMENT$	Count	LS Mean	LS Sigma	Homogeneous Groups
Lockport	64	2.82813	0.357261	X
Brandon	104	5.12474	0.280524	X
Upper Dresde	72	9.15315	0.33729	X
Lower Dresde	87	10.427	0.306685	X

Contrast	Difference	+/- Limits
Brandon - Lockport	*2.29662	0.893645
Brandon - Lower Dresde	*-5.30225	0.818434
Brandon - Upper Dresde	*-4.02841	0.864046
Lockport - Lower Dresde	*-7.59887	0.926315
Lockport - Upper Dresde	*-6.32502	0.966614
Lower Dresde - Upper Dresde	*1.27384	0.895897

* denotes a statistically significant difference.

(B) Multiple Range Tests for SUNFISH by Reach

Method: 95.0 percent LSD

SEGMENT$	Count	LS Mean	LS Sigma	Homogeneous Groups
Lockport	64	0.578125	0.108376	X
Brandon	104	1.10639	0.0850973	X
Upper Dresde	72	1.95743	0.102317	X
Lower Dresde	87	2.63009	0.0930332	X

Contrast	Difference	+/- Limits
Brandon - Lockport	*0.528269	0.271088
Brandon - Lower Dresde	*-1.5237	0.248273
Brandon - Upper Dresde	*-0.851037	0.262109
Lockport - Lower Dresde	*-2.05197	0.280999
Lockport - Upper Dresde	*-1.37931	0.293223
Lower Dresde - Upper Dresde	*0.672663	0.271771

* denotes a statistically significant difference.

Table 6.6 Multiple Comparisons Test for Percentage of fish by Reach in the Lower Des Plaines River that are (A) Tolerant Species and (B) Exhibiting DELT Anomalies.

(A) Multiple Range Tests for TOLERANTPCT by REACH

Method: 95.0 percent LSD

SEGMENT$	Count	LS Mean	LS Sigma	Homogeneous Groups
Lower Dresde	87	21.6966	2.49496	X
Lockport	60	30.4869	3.002	X
Upper Dresde	72	35.1703	2.74396	X
Brandon	103	44.0048	2.29269	X

Contrast	Difference	+/- Limits
Brandon - Lockport	*13.5179	7.43386
Brandon - Lower Dresde	*22.3082	6.67196
Brandon - Upper Dresde	*8.83452	7.04215
Lockport - Lower Dresde	*8.79033	7.67765
Lockport - Upper Dresde	-4.68338	7.99891
Lower Dresde - Upper Dresde	*-13.4737	7.28868

* denotes a statistically significant difference.

(B) Multiple Range Tests for DELTPCT by REACH

Method: 95.0 percent LSD

SEGMENT$	Count	LS Mean	LS Sigma	Homogeneous Groups
Lower Dresde	87	5.3985	1.04177	X
Lockport	60	7.98223	1.25348	XX
Brandon	103	9.00612	0.957311	X
Upper Dresde	72	11.1713	1.14574	X

Contrast	Difference	+/- Limits
Brandon - Lockport	1.02389	3.10401
Brandon - Lower Dresde	*3.60763	2.78588
Brandon - Upper Dresde	-2.16521	2.94045
Lockport - Lower Dresde	2.58374	3.2058
Lockport - Upper Dresde	-3.1891	3.33994
Lower Dresde - Upper Dresde	*-5.77284	3.04339

* denotes a statistically significant difference.

Comparison to Reference Sites in Illinois

One of the major advantages of using a multimetric IBI for characterizing fish communities and their response to stressor gradients is that multimetric indices are constructed and calibrated by comparing large numbers of fish communities encompassing a wide range of impact levels. Because the Ohio Boatable River IBI was not constructed specifically to describe variation among fish communities in Illinois, the Biological Subcommittee agreed that it was necessary to compare the IBI values calculated for the Lower Des Plaines River with IBI values calculated for other rivers in Illinois that were known to differ in their levels of human-induced impacts.

Three sets of data were identified for use in this analysis, including stations on the Green River, Rock River and Fox River. The Green and Rock River are considered to have relatively low levels of human impact, whereas the Fox River has higher levels of degradation due to impoundments, water quality, and legacy chemical effects. In addition, data from the Upper Des Plaines River were included in the comparison. Although the Upper Des Plaines is physically and hydrologically quite different from the Lower Des Plaines, it can provide a useful reference for potential sources for fish migrating into the Lower Des Plaines River. Data for the Green, Rock and Des Plaines were provided by the Illinois EPA and data for the Fox River were provided by US EPA. All data used were collected using boat mounted electrofishing gear. Levels of sampling effort differed among stations in both station lengths and time spent sampling, unlike sampling procedures outlined for the application of the Ohio IBI (Ohio EPA, 1988). Due to different station lengths, fish abundances were adjusted to a common distance for analysis, but low-end corrections were not applied. DELT anomalies were not recorded for the Green, Rock, and Upper Des Plaines River stations, and as such an average score of 3 was used.

A comparison of the Ohio IBI scores calculated for the reference sites with those calculated for reaches of the Lower Des Plaines are presented in Figure 6.7. Three major points can be made from this analysis. First, the range of IBI scores among the reference sites confirms that the Ohio IBI does capture the range of degradation among Illinois rivers that was predicted a priori by the Biological Subcommittee; the Green and Rock Rivers scored high (median IBI > 40), while the free-flowing reaches of the Fox River scored intermediate (median IBI ~ 32) and the impounded reaches of the Fox River scored lowest (median IBI ~ 21). Second, the large and significant difference in IBI between the impounded and free-flowing stations of the Fox River make a strong case that the habitat modifications resulting from pooling of water behind dams results in major declines in biotic integrity, independent of other interacting watershed-related factors. Third, IBI scores for Upper and Lower Dresden stations are comparable to those for the impounded reaches of the Fox River, whereas the IBI scores for Brandon and Lockport Pools are significantly lower than the impounded Fox River sites.

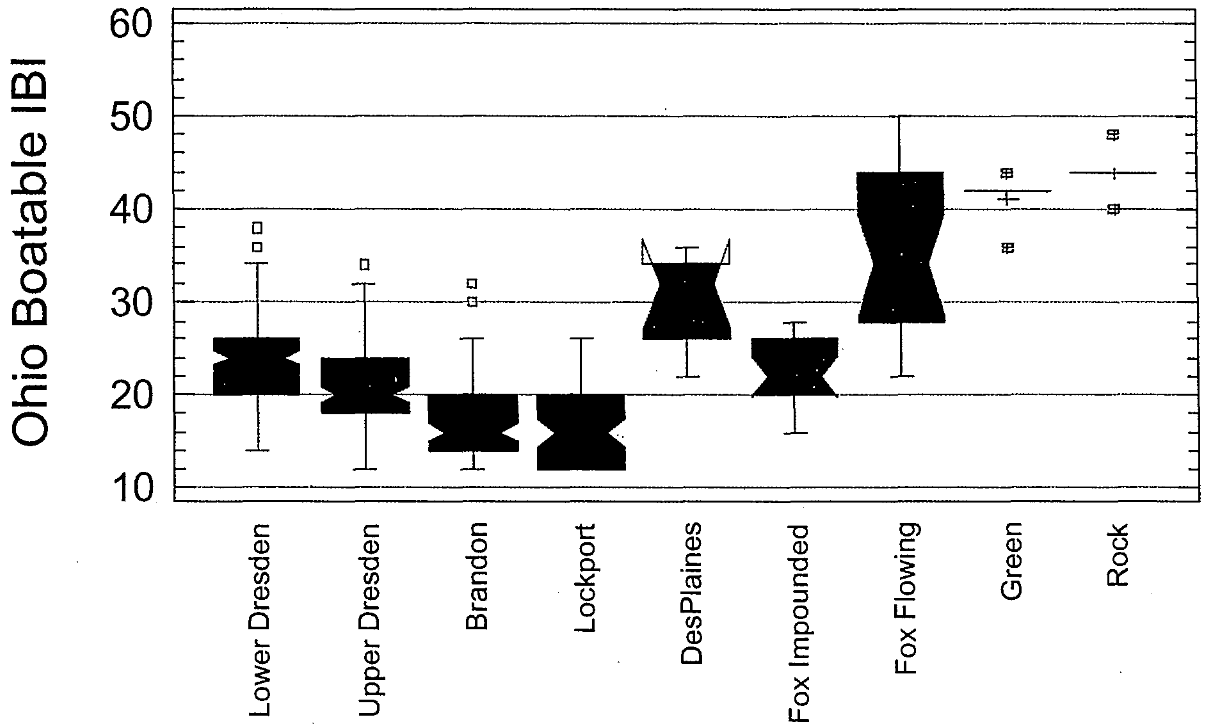


Figure 6.7 Comparison of Ohio IBI scores calculated for stations in the Lower Des Plaines Waterway and selected sites in Illinois.

Analysis of Individual Metrics Contributing to IBI Scores for Reference Sites

Examination of the component IBI metrics among reference streams (Figures 6.8-6.10) further confirms the patterns of degradation in the Lower Des Plaines system. The numbers of native species (NATIVE) and sucker species (SUCKER) exhibit higher levels in the Green, Rock, and free-flowing sections of the Fox River compared to the Lower Des Plaines and impounded reaches of the Fox River (Figure 6.8). The number of sunfish species (SUNFISH) is generally higher in the reference streams than in the more impacted reaches of the Lower Des Plaines, but the large variance among samples makes drawing any statistically-based conclusions difficult. A similar, but opposite, pattern is observed in the percentage of fish that are omnivorous (OMNIVPCT, Figure 6.10), with the reference streams having lower levels of omnivores compared to the Lower Des Plaines. It is worthy to note that intolerant species (INTOLERANT) are absent from all sites in the Lower Des Plaines, but exhibit an increase in abundance among reference sites in general correlation with the a priori hypothesized degradation gradient (Figure 6.8). Similar trends can be seen in the percent of round bodied suckers (RBSKRSPCT, Figure 6.9) and percent simple lithophilic species (LITHOPHPCT, Figure 6.10). There is a noticeably higher variance in simple lithophiles in the Upper Dresden and Brandon Pool reaches, due in part to the inclusion of dam tailwater habitats which typically contain rocky substrates and faster-flowing water preferred by lithophiles.

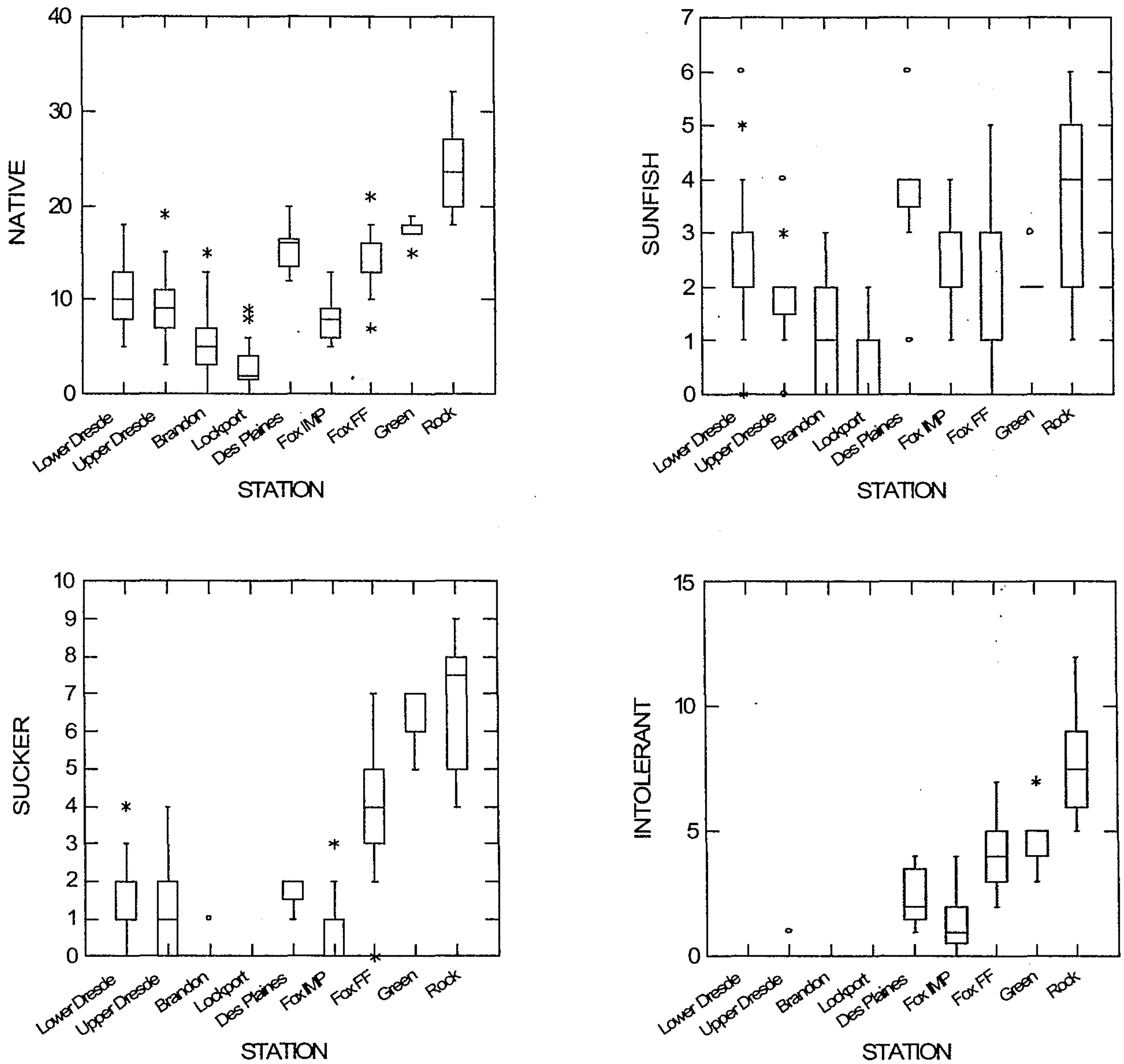


Figure 6.8 Examination of Contributing Metrics used to calculate Ohio Boatable IBI for Des Plaines River and Reference Stations.

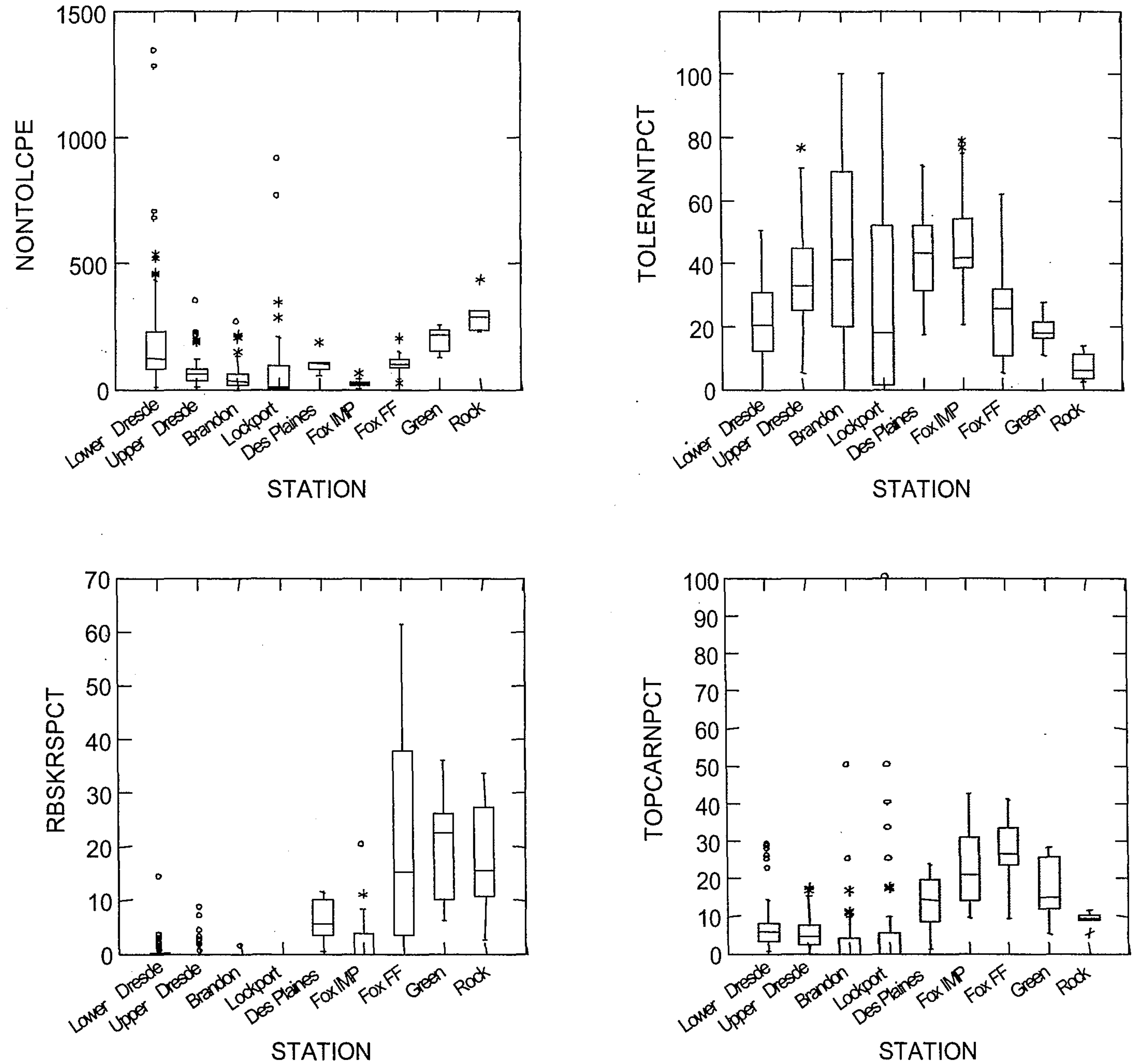


Figure 6.9 Examination of Contributing Metrics used to calculate Ohio Boatable IBI for Des Plaines River and Reference Stations.

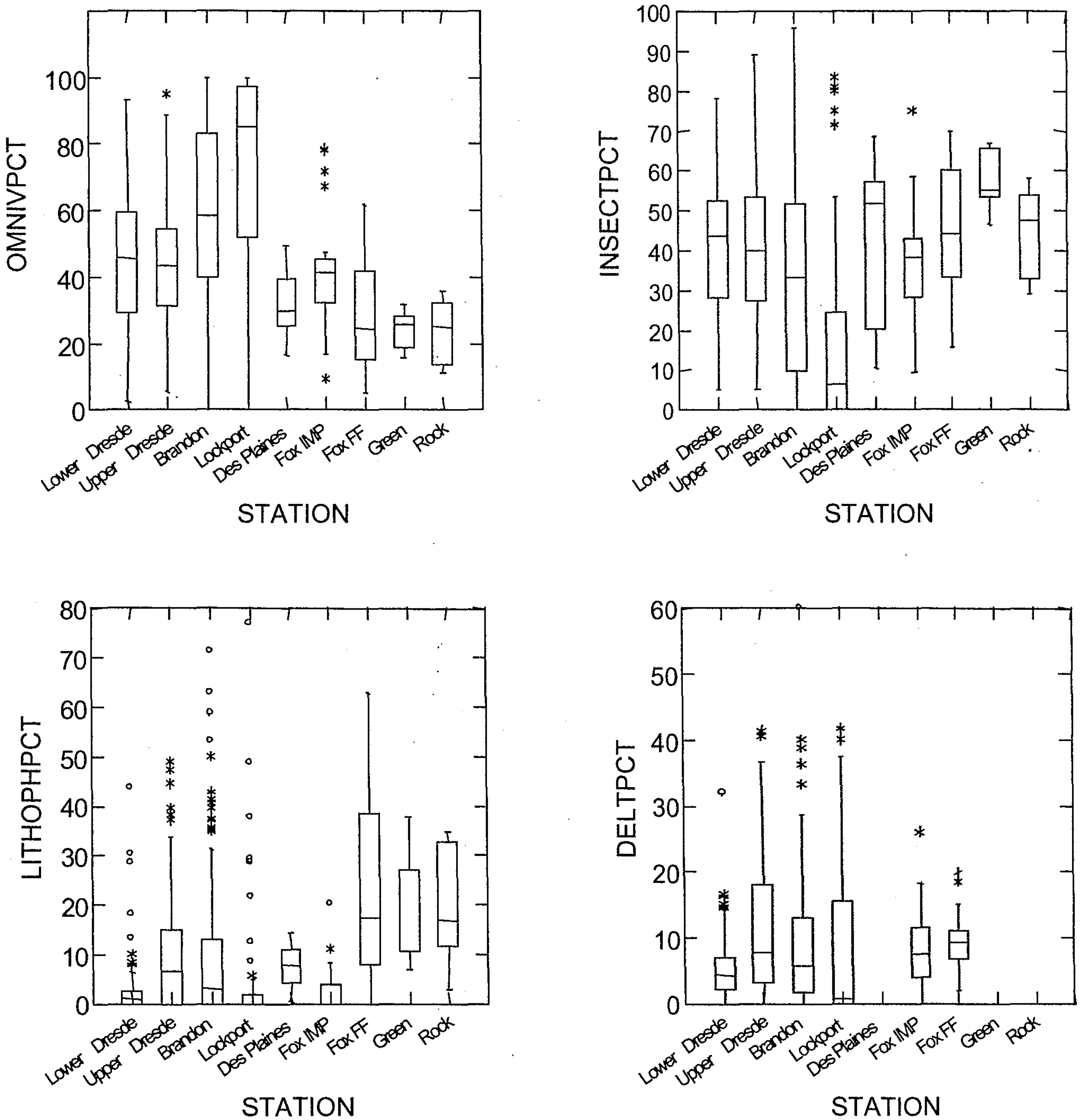


Figure 6.10 Examination of Contributing Metrics used to calculate Ohio Boatable IBI for Des Plaines River and Reference Stations.

Stresses on the Biota

Habitat

Habitat was characterized both by habitat type (Backwater, Main Channel Border, Main Channel, Tributary Mouth, and Tail Water) and by QHEI (see Chapter 4). The relationship between habitat type and QHEI for each of the four reaches is shown in Figure 6.11. QHEI values are higher in Lower and Upper Dresden, largely due to the absence of back water and Tributary Mouth habitats in Brandon and Lockport Pools.

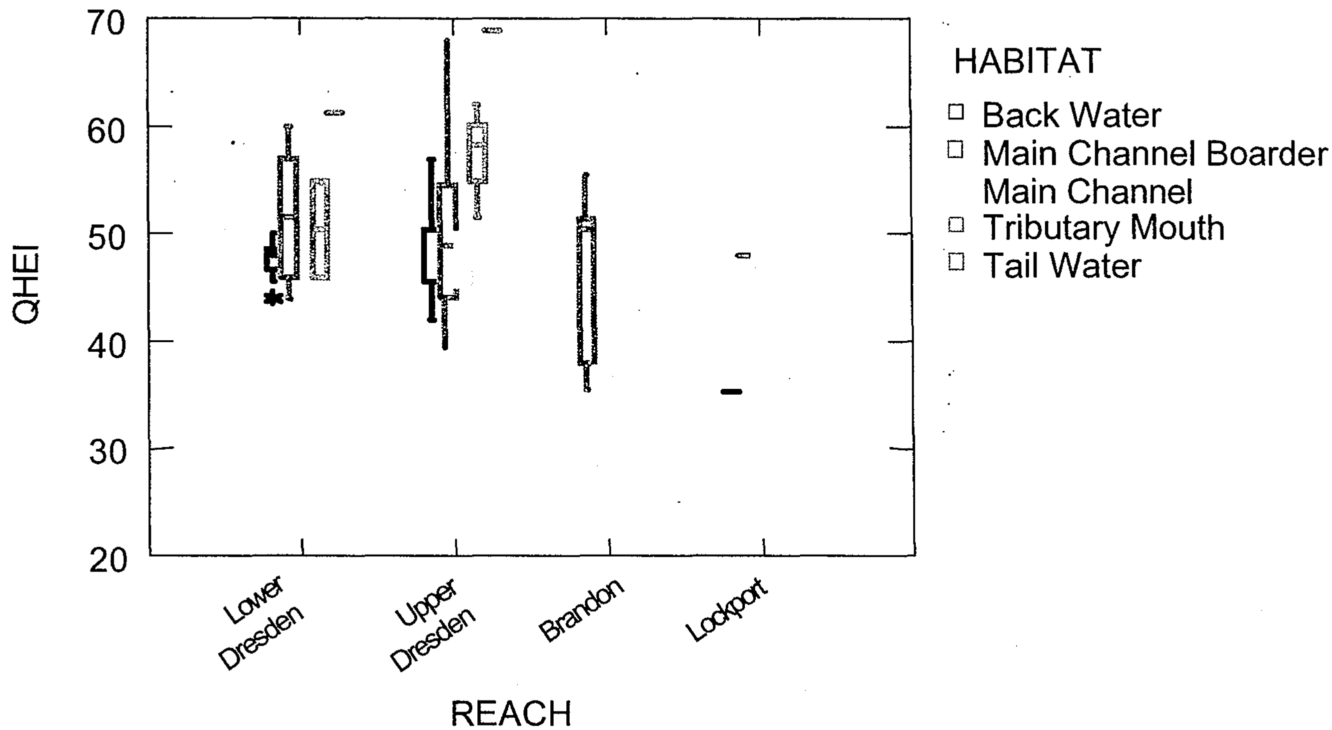


Figure 6.11 QHEI Scores pooled by habitat type for reaches within the Lower Des Plaines River and CSSC.

An Analysis of Covariance (ANCOVA) with main effects of Reach, Habitat Type, and Month, (using QHEI as a covariate) was conducted for IBI scores from 2000 (the same year that QHEI data were collected). This analysis has the effect of answering the question as to whether the differences in IBI among reaches can be accounted for due to differences in Habitat Type, QHEI, or Month when sampling occurred. The results of the analysis are presented in Table 6.7 The analysis shows significant effects of Month, Habitat Type, and Reach, but no significant relationship with QHEI (Table 6.7). This further supports the contention that the lack of tributary mouth, tail water, and back water habitat types in Brandon and Lockport contribute to the lower IBI scores.

Table 6.7 Analysis of Covariance Variance for the effects of Habitat (QHEI) on IBI, with main effects of Month, Habitat Type, and Reach. IBI data were used from 2000, the same year that QHEI data were collected.

Analysis of Variance for IBI - Type III Sums of Squares

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
COVARIATES					
QHEI	0.749266	1	0.749266	0.05	0.8298
MAIN EFFECTS					
A:MONTH	851.89	4	212.972	13.20	0.0000
B:HABITAT$	243.665	4	60.9162	3.78	0.0065
C:REACH$	175.258	3	58.4192	3.62	0.0155
RESIDUAL	1710.25	106	16.1344		
TOTAL (CORRECTED)	3545.28	118			

Seasonal Impacts

Although there is a consistently significant effect of sampling month on IBI scores, the direction and magnitude of the effect varies among reaches and between years. Figure 6.12 illustrates these patterns for 2000 and 2001. Median IBI for Lockport and Brandon Pools are always lower than for upper and lower Dresden pools. However, even though on average lower Dresden has a higher IBI than upper Dresden, there are months when this pattern is reversed. This reversal in scores suggests that the factor(s) responsible for the general decline moving upstream from lower to upper Dresden may not exert consistent stress across time. This further indicates that the responsible factor is most likely is not habitat (which remain relatively constant over the summer months), but more likely a temporally dynamic factor like temperature which is more variable and which may also serve as a barrier to fish movement between reaches.

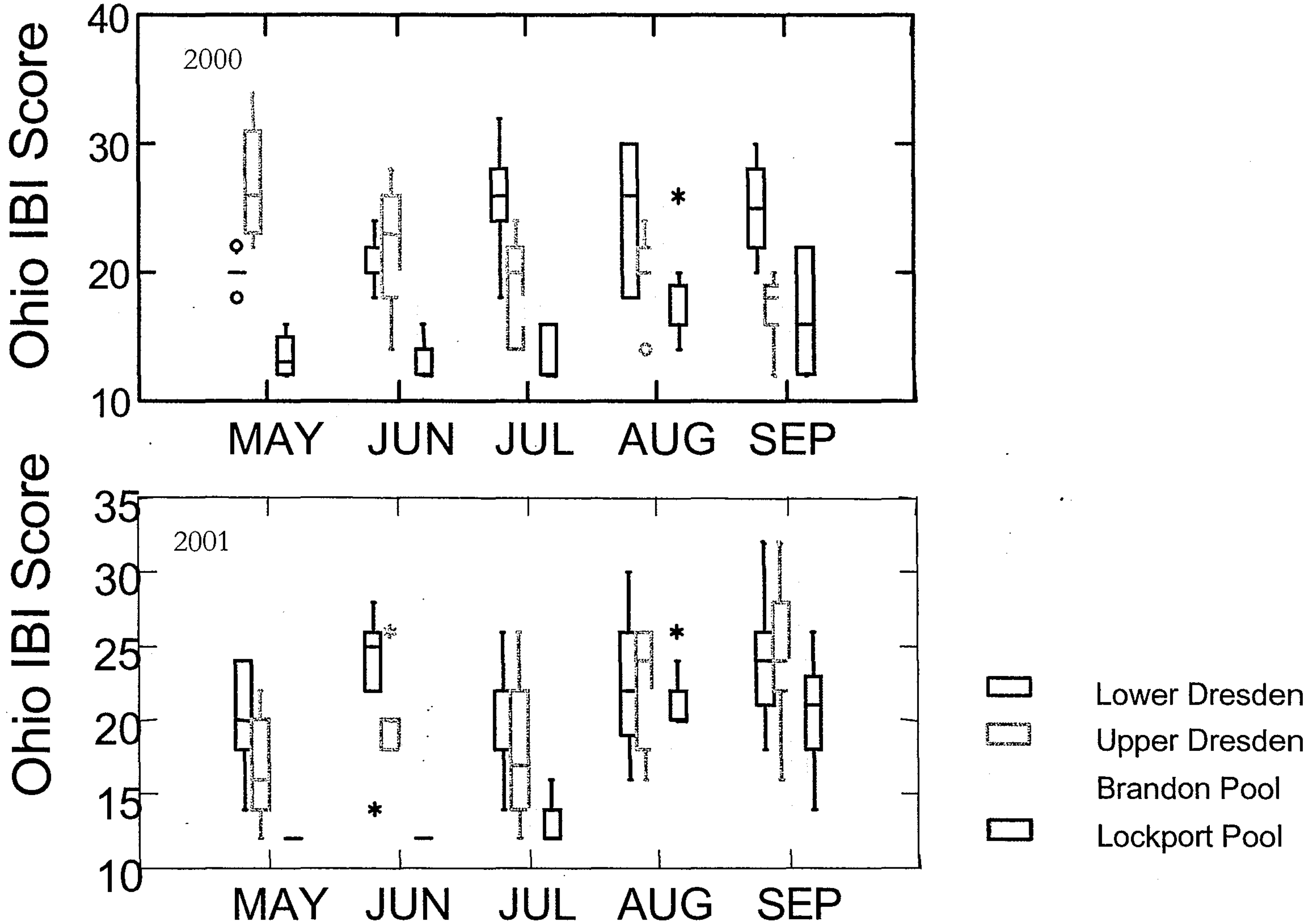


Figure 6.12 Monthly changes in IBI Scores pooled by habitat type for reaches within the Lower Des Plaines River and CSSC for 2000 and 2001.

Summary – Potential Fish Community

The analysis of data for the Lower Des Plaines River and comparison of this data with collections from other Illinois rivers indicates that the Ohio IBI Boatable River multimetric index is an appropriate tool for characterizing the status and trends in fish communities in impacted waters of Illinois. Assessment of IBI scores reveals a statistically significant decrease in biotic integrity moving upstream from Lower Dresden, to Upper Dresden, and into Brandon Pool.

IBI scores for Upper and Lower Dresden are not significantly different than those for impounded reaches of the Fox River. However, free-flowing reaches of the Fox River have significantly higher IBI scores. This suggests that the presence of and proximity to dams has significant effects on the fish biotic integrity.

Examinations of some component metrics of the IBI suggest that the underlying stressors may be expressed along a continuous spatial gradient (e.g. numbers of native species and percent tolerant species in Figure 6.6). Factors such as high temperatures, low dissolved oxygen, water quality degradation, or loss of habitat could produce such trends. On the other hand, several other metrics suggest Reach-specific factors (e.g. numbers of sucker species in Figure 6.6) which could be produced by factors such as legacy sediment contamination or barriers to fish passage such as dams.

Analysis of covariance methods show that most of the difference in IBI scores between Upper and Lower Dresden for samples collected in 2000 can be accounted for by seasonal effects and differences in the availability of types of habitat (Table 6.7). Even after accounting for these effects, Brandon Pool still has lower IBI scores compared to Upper and Lower Dresden. The significant Habitat Type effect suggests that habitat improvement in Upper and Lower Dresden could result in improvement of fish communities. However, the significant month effect raises the possibility that either temperature and/or oxygen are potential factors that could be responsible for some of the observed patterns detected in the data.

Base on the Water Body Assessment in Chapters 2 and 3, DO conditions are critical in the Brandon Road pool (both low mean DO and daily variations) and high temperature and daily DO fluctuations are preventing the attainment of the use in the Dresden Island Pool between the thermal discharges and I-55.

Analysis in Chapter 3 also indicated a problem with the legacy pollution contained in the sediments that can have a chronic effect on the food chain, beginning with benthic invertebrates, and could propagate to fish.

The State of Ohio has developed a set of IBI criteria for determining compliance with the goals of the Clean Water Act. The criteria are developed for stream type and reflect stream modifications such as "Channel Modification" and "Impounded". The numerical criteria are based on sampling conducted at more than 350 reference sites that typify the "least impacted" condition within each of the states five Ecoregions (Yoder and Rankin, 1995). For the Eastern

Corn Belt Plains Ecoregion, the State of Ohio has established the following IBI criterion for boatable waters:

Warmwater Habitat	48
Impounded	30
Channel Modified	24

The Ohio "Warmwater Habitat" stream classification would correspond to the Illinois "General Use" classification. None of the Lower Des Plaines River reaches studied meet the Warmwater Habitat criterion. The Lower Dresden Pool comes close to meeting the channel-modified criteria with a mean value of 23.79. The Upper Dresden Pool is at 20.51 and is below the channel-modified criterion. The Brandon Pool at 17.40 and Lockport Pool at 16.45 fall far below the Ohio numerical IBI criteria for channel modified streams.

As discussed above, part of the reason for the poor IBI values throughout the Lower Des Plaines River is lack of adequate habitat. While artificial improvements in habitat could raise IBI scores, as discussed in Chapter 4 of this report, habitat improvement opportunities in the Brandon Road Pool are limited by the maintenance of the federal navigation channel. While habitat improvement opportunities exist in the Dresden Island Pool, the improvements are limited to improvements in riparian habitats. Introduction of substrate diversity and riffle habitats is difficult in the entire Lower Des Plaines River due to the impounded condition of the river. Meeting an IBI value of 48 for "warmwater habitat" does not appear feasible because of the artificial modifications to the stream channel.

However, there are significant temperature (Dresden Island pool) and DO (both Brandon and Dresden Island) stresses. Removing these stresses would bring about marked improvement of the water quality and biotic integrity conditions in both pools. These impediments to the attainment of water quality should be remedied. The effect of the contaminated sediments on fish population is less clear and will require further study.

Box 1.1 (Chapter 1) outlines the six reasons for a change of the designated use of a water body as outlined in Federal Regulation 40 CFR 131. Reason number 4 for a change of the designated use and/or water quality standards for a water body states:

> *"Dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use"*

Based on reason number 4, it is recommended that the entire Lower Des Plaines River, including the Branden Road and Dresden Island Pools, be considered for a modified stream classification that would reflect the currently altered habitat of the waterway.

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CHAPTER 7

PATHOGENS AND RECREATION

The Lower Des Plaines River is not a recreational water body. Its major uses are navigation and wastewater conveyance. It receives sewage, industrial wastewater discharges, and urban runoff from the Chicago metropolitan area (population about 9.5 million). However, the river flows through a major urban center - the City of Joliet - and has significant aesthetical assets and occasional use of the river is a possibility. Downstream of the I-55 bridge, the current boundary between the Illinois General Use and Secondary Contact and Indigenous Life Use, the primary and secondary recreational use is more wide spread. Consequently, the Use Attainability Analysis must address the question of protection for swimmers and recreationists as required by the Clean Water Act and Water Quality Standards regulations (40 CFR 131). Currently, no microbiological standards are in force for the river between the Lockport Lock and Dam and the I-55 bridge.

Review of Current Limits

Illinois General Use

a) During the months of May through October, based on a minimum of five samples taken over not more than a 30 day period, fecal coliforms (STORET No. 31616) shall not exceed a geometric mean of 200 per 100 mL, nor shall more than 10% of the samples during any 30-day period exceed 400 per 100 mL in protected waters. Protected waters are defined as waters which, due to natural characteristics, aesthetic value or environmental significance are deserving protection from diseases caused by pathogenic organisms. Protected waters will meet one or both of the following conditions:

   1) presently support or have the physical characteristics to support primary contact; and/or
   2) flow through or adjacent to parks or residential areas.

b) Waters unsuited to support primary contact uses because of physical, hydrologic or geographic configurations and are located in areas unlikely to be frequented by the public on a routine basis as determined by the Agency at 35. Ill. Adm. Code 309 Subpart A, are exempt from this standard.

Illinois Secondary Use

The Illinois Pollution Control Board in 1972 (IEPA, 1972) adopted a standard for *Restricted Use Waters.* Restricted use meant that certain uses were not protected. The restricted use extended for the entire CSSC and the Lower Des Plaines River to the I-55 bridge (RM 278). Later the "restricted use" was renamed as the Secondary Contact and Indigenous Aquatic Life Use. The restricted use standard for bacteria was:

> Based on a minimum of five samples taken over not more than a 30 - day period, fecal coliforms shall not exceed a geometric mean of 1000/100 ml, nor shall more than 10% of samples taken during any 30-day period exceed 2000/100 mL.

The above previous standard was repealed effective October 26, 1982 and no standards for bacterial pollution are today in force for the secondary contact (restricted) use.

Federal Water Quality Criteria

Original Formulation (Water Quality Criteria, USEPA 1986)

The federal criteria for microbiological freshwater pollution were included in the USEPA (1986) criteria publication. The criteria were formulated as follows:

> Based on a statistically sufficient number of samples (generally not less than 5 samples equally spaced over a 30-day period), the geometric mean of the indicated bacterial densities should not exceed one or the other of the following:

E.coli	126 per 100 mL; or
enterococci	33 per 100 mL;

> no sample should exceed a one - sided confidence limit (CL) calculated using the following guidance:

designated bathing beach	75% CL
moderate use for bathing	82%
light use for bathing	90%
infrequent use for bathing	95%

> based on a site specific log standard deviation, or if site data are insufficient to establish a log standard deviation, then using 0.4 as the log standard deviation for both indicators.

States should adopt both the geometric mean and the single maximum (based on the expected frequency of bathing) criteria into their water quality standards to protect public beaches. The single maximum should be used for the designated bathing areas (USEPA, 2000). This dual criterion should also be used in preparation of the 305(b) reports (USEPA, 1997). For the 303(d) listing leading to the TMDL (if the standard is not met) the geometric mean as well as the simple sample maximum, regardless of the number of samples taken, determine compliance or noncompliance with the standard. The minimum number of samples (in the 30-day period) is specified for accuracy purposes (USEPA, 2000).

USEPA Guidelines to Implement the Criteria for Recreation

The views of the USEPA are expressed in several key documents. First, the USEPA (1986) criteria document specifies the magnitude of the criterion for the two indicator organisms (escherichia coli and enterococci) but is not specific as to the frequency dimension of the criterion. The second document, the USEPA (1994) criteria handbook, presents and discusses options how to designate the primary and secondary contact recreation uses for water bodies. The third document is the water quality standard regulation contained in 40 CFR 131 and its draft modification published in the July 7, 1998 *Federal Register*. The fourth document is the draft implementation guidance document contained in the USEPA (2000, 2002) documents that reiterate the use of the 1986 criterion based on the use of *Escherichia coli* and *enterococci* indicator organisms. A brief discussion of the most important rules and guidance is presented herein.

Selection of Designated Use

Water Quality Standards Handbook (USEPA, 1994)

The handbook provides extensive suggestions on selection of primary and secondary contact recreation. The handbook, which apparently is a guidance document that is not legally binding, defines the primary and secondary recreation use classifications as follows:

> *Primary contact recreation usually includes swimming, water skiing, skin-diving, surfing, and other activities likely to result in immersion.*
>
> *The secondary contact classification is protective when immersion is unlikely. Examples are boating, wading, and rowing. Fishing is often considered in the recreational use categories.*

The guidelines contained in the USEPA (1994) handbook for establishing the standards for recreational use were stringent. Essentially, the handbook stated that primary contact recreation is a mandatory use for all (navigable) water bodies unless a UAA proves that the use is not attainable. However, using irreversible physical deficiencies that prevent the use as a reason to remove the primary contact use was disallowed. The book outlined two options, both requiring an adoption of primary contact recreation standards. These stringent requirements were subsequently relaxed in the draft USEPA (2000, 2002) guideline documents.

Indicator Organisms - The Need for Change

In relation to the current Illinois General Use standards for recreation, the most important issue is the difference in the choice of indicator microorganisms. The Illinois General Use Standard use fecal coliforms as indicator organisms and the USEPA is urging in USEPA (2000, 2002) documents use of the *E. coli* and/or *enterococci* indicator microorganisms.

The change from fecal coliforms to *E. coli*/enterococci indicators represents a shift in philosophy for the protection of swimmers against gastrointestinal and other diseases that may occur by ingesting or contacting water contaminated by pathogens. Before 1986, USEPA and almost all states were using fecal coliforms as the indicator organisms. Fecal coliforms criteria and state standards were perceived as protecting swimmers from waterborne infectious gastrointestinal and other diseases caused by fecal pollution, primarily of human origin. Bacteria of the fecal coliform group are considered to be the primary indicators of fecal contamination because they are associated in high numbers with the gastrointestinal tracks and feces of humans and warm-blooded animals. They are also present in the digestive tracks in quantities that far exceed other pathogens. Bacterial pollution constitutes a health risk to both swimmers and recreationists on and in the water and also can contaminate shellfish. Cabelli (1977) found that, among swimmers, the most significant illness was an acute, relatively benign form of gastroenteritis. However, recent outbreaks of illnesses associated with *E. coli* (mostly for eating insufficiently cooked contaminated meat) and waterborne sickness caused by *cryptosporidium* heightened the concerns with waterborne pathogens.

The reason for the change of the indicator organisms is apparently the fact that gastrointestinal sicknesses have occurred even when swimmers were in contact with the water that met the standard expressed by the fecal coliforms indicator (e.g., Seyfried *at al.,* 1985; Calderon et al., 1991). Microorganisms of the coliform group of both human and nonhuman fecal and nonfecal origin have been found to cause such disease. In addition, other organisms that can be both of the fecal (human and animal) and non fecal origin can be pathogenic, such as *cryptosporidium, streptococci, or staphylococci* (Seyfried *et al.,* 1985). Therefore, USEPA (1986, 2000, 2002) concluded that for fresh water bodies, *escherichia coli (E. coli)* and *enterococci* are best suited for predicting the presence of gastrointestinal illness causing pathogens, and *enterococci* is best suited for marine beaches.

The USEPA (1988) compendium of state standards for bacteria documented that the great majority of states were using exclusively fecal coliforms as an indicator of pollution by pathogenic organisms. In 1999, only 16 states adopted the 1986 E. coli/enterococci indicators (USEPA, 2000). The remaining states are still using the fecal coliforms indicators, including the State of Illinois. The USEPA (2000, 2002) draft guidance document encourages states to make the transition from fecal coliforms to E. coli/enterococci indicator organisms and bacterial contamination testing during this triennial review of the state standards. *In the draft document* ***USEPA (2000) states that if a State, Territory, or authorized Tribe does not adopt USEPAís recommended 1986 bacteria water quality criteria during this period, EPA intends to act under Section 303(c)(4)(B) of the Clean Water Act (CWA) to promulgate federal water quality standards, with the goal of assuring that EPAís recommended 1986 water quality criteria apply in all States, Territories, and authorized Tribes, as appropriate, by 2003.***

USEPA's (2002) latest draft criteria implementation document is a revision of the previously issued USEPA (2000) guidance. The 2002 guidance provides states with more flexibility in developing and defining the standards for recreation and bacterial contamination. The key feature is that these criteria are risk based where the risk of getting a waterborne gastrointestinal disease is the primary criterion to which the numeric numbers of E.coli and enterococci microorganisms are correlated using data of epidemiological studies of bathers. Thus, the state has more options for establishing the standards based on probability of contact recreation of the water body in question.

The USEPA (2002) guidance tightened the schedule for the implementation of change from fecal coliforms to E. coli by the states. The USEPA now requires that states adopt the new criteria as standards either immediately or within a three year transitional period during which both (old) fecal coliforms standards and (new) *E.coli* standards are in force. After the three-year transition the new standards should be fully implemented. It is recommended that *E.coli* indicators are used for fresh water and E.Coli and enterococci for marine waters. The necessity to switch to the *E. coli* indicator was highlighted by the new Section 303(i) if the Clean Water Act which requires coastal states to adopt new or revised water quality standards for pathogenic microorganisms and pathogen indicators by April 10, 2004. This amendment, called Beaches Environmental Assessment and Coastal Health

Act (BEACH Act) was passed by Congress on October 10, 2000. The BEACH Act Amendment also directs the USEPA to promulgate such standards for states that fail to do so. In general, Great Lakes coastal waters would fall under this amendment and Illinois is a coastal state.

Standards Linked to Risk of Illnesses

When the US EPA published its criteria in 1986, the criteria were based on the illness rate (risk) of 8 illnesses per 1000 swimmers for fresh waters. A higher rate was adopted for marine waters. This rate of illness was commensurate to the previous fecal coliform criterion. **The current guidelines allow states to adopt criteria, based on the frequency of uses of the water body for swimming, for illness rates from 8 illnesses/1000 swimmers to 14 illnesses /1000 swimmers**. The low illness rate corresponds to highly frequented beaches. Table 7.1 presents the *E. coli* criteria expressed in colony forming units (cfu[1]) per / 100 mL related to the risk of gastrointestinal illness. The table contains the criterion for the geometric mean of five samples taken over a period of 30 days, and single sample maximum. The maximum value for a single sample is calculated from a log-normal probability distribution of the samples with a logarithmic standard deviation of 0.4.

Table 7.1 E. Coli Criteria (USEPA, 2002) for Primary Contact Recreational Use

Illness Rate (per 1000)	Geometric Mean Density cfu/100mL	Single Sample Maximum Allowable Density (cfu/100 mL)			
		Designated Beach Area 75% C.L.	Moderate Full Body Contact Recreation 82% C.L.	Lightly Used Full Body Contact 90% C.L.	Infrequently Used Full Body Contact 95% C.L.
8	126	235	298	408	576
9	160	300	381	524	736
10	206	383	487	669	941
11	263	490	622	855	1202
12	336	626	795	1092	1536
13	429	799	1016	1396	1962
14	548	1021	1298	1783	2507

Use of the criteria for pathogens requires the use of statistics. Among other reasons, this is because the limiting values in Table 7.1 were calculated using the logarithmic standard deviation of 0.4. If the probabilistic distribution of measured data yields a standard deviation statistically greater than 0.4, the distribution is such that while the geometric mean limit is consistently met for the particular water body, the single value maximum would be routinely

[1] Cfu means *colony forming units* when a more common membrane test for bacteria is used.

exceeded. In this case, as described in the USEPA (1986) *Ambient Water Quality Criteria for Bacteria* and reconfirmed in the USEPA (2002) draft guidance, <u>a state may re-calculate a standard deviation specific to the water body and subsequently adopt into water quality standards single sample maximum values specific to the observed distribution of criteria.</u>

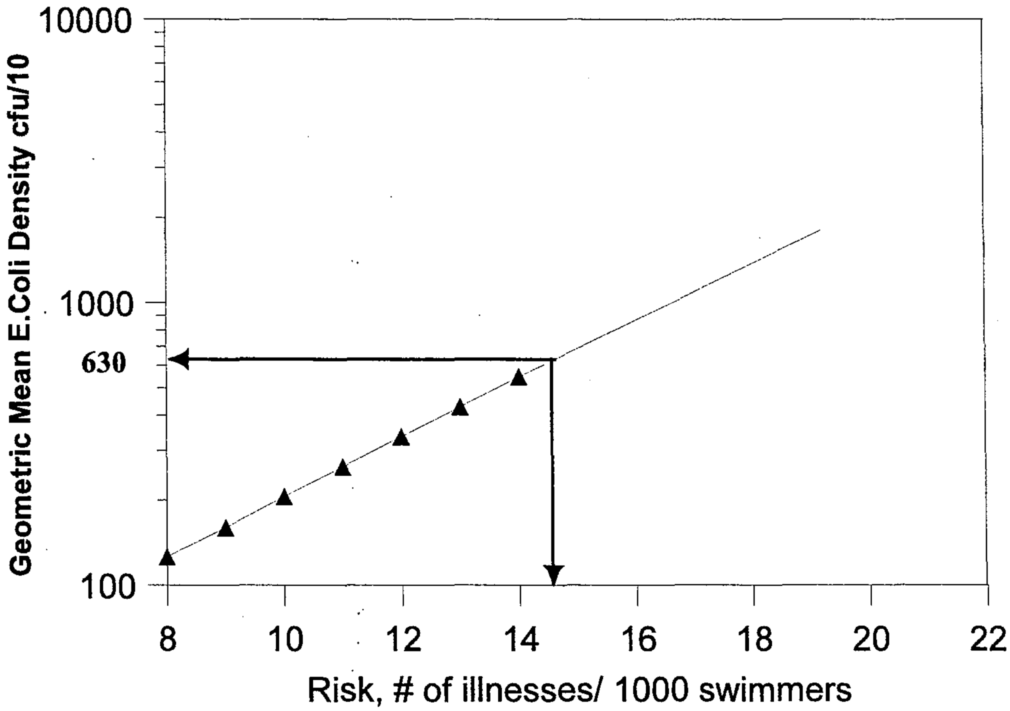

Figure 7.1 **Relation of the E. coli standard to the risk of contacting waterborne illness. The relation beyond the risk of 14 illnesses/1000 swimmers was extrapolated for USEPA (2002) guidelines data.**

Figure 7.1 is a graphical representation of the relation of the risk of contacting an illness to the density of *E. coli* in water. The relationship is semi logarithmic, i.e., the logarithm of the density plotted as a straight line against the arithmetic risk.

This chart enables the extrapolation of the risk to higher bacterial densities.

Use of a single maximum value is mostly used by beach managers. The geometric mean value is the criterion that is more appropriate for long term evaluation and TMDL planning.

Boxes 7.1 and 7.2 shows examples of approved standards for waterborne recreation by Colorado and Ohio. Both states defined standards for primary and secondary contact recreation. Implementing secondary recreation standards requires a UAA. The selection of the risk between the ranges of 8 to 14 illnesses/1000 people is a management and policy decision by the IEPA, similar to the

Box 7.1 Bacteria standards for Colorado

Colorado has two categories of primary contact recreation use in addition to their secondary designated use. The Recreation Class 1A use is the default use category, and is assigned an E. Coli criterion (standard) of 126 cfu/100 mL based on the EPA recommended risk of 8 illnesses per 1000 swimmers. The recreation 1B use is intended to protect waters with the potential to support primary contact recreation use but it can be assigned only if a reasonable level of inquiry has failed to identify any existing primary contact recreation uses of the water body. This use category is assigned the standard of 206 cfu/100 mL, commensurate to the risk of 10 illnesses per 100 swimmers. The secondary recreation use may be assigned in Colorado only where a use attainability analysis has been conducted that further demonstrates there is no reasonable potential for primary contact recreation uses to occur within the next 20 years. This use category is assigned the geometric mean *E. coli* criterion of 630 cfu/100 mL that equals five times the geometric mean value associated with 8 illnesses per 1000 swimmers.

Box 7.2 Ohio Standards for Recreation

Recreation standards for the state of Ohio represent another example of successful adoption of federal criteria. The interesting feature of the standards is the shift in the interpretation of contact recreation and secondary contact use. Ohio divided the primary contact use into two categories and redefined the secondary contact. The state developed the standards using both fecal coliform and E. coli indicator organisms. *At least one (not both) of the two bacteriological standards must be met.* The recreation use designations and standards are:

(A) *"Bathing waters"* - these are waters that, during the recreation season, are suitable for swimming where a lifeguard and/or bathhouse facilities are present.

Standards: **Fecal coliforms** - geometric mean FC content, based on not less than five samples within a thirty day period, shall not exceed 200/100 mL and fecal coliform content shall not exceed 400/100 mL in more than ten percent of the samples taken during any thirty - day period.

E. coli - geometric mean E . coli content, based on not less than five samples within a thirty day period, shall not exceed 126/100 mL and E. coli content shall not exceed 235/100 mL in more than ten percent of the samples taken during any thirty - day period.

Box 7.2 Ohio Standards for Recreation - Continuing

(B) *"Primary contact"* - these are waters that, during the recreation season, are suitable for full- body contact recreation such as, but not limited to, swimming, canoeing, and scuba diving with minimal threat to public health as a result of water quality. *All lakes and reservoirs, except upground storage reservoirs and those lakes and reservoirs meeting the definition of bathing waters, are designed primary contact recreation.*

Standards: **Fecal coliforms** - geometric mean FC content, based on not less than five samples within a thirty day period, shall not exceed 1000/100 mL and fecal coliform content shall not exceed 2000/100 mL in more than ten percent of the samples taken during any thirty - day period.

E. coli - geometric mean E . coli content, based on not less than five samples within a thirty day period, shall not exceed 126/100 mL and E. coli content shall not exceed 298/100 mL in more than ten percent of the samples taken during any thirty - day period.

(C) *"Secondary contact"* - these are waters that during the recreation season, are suitable for partial body contact recreation such as, but not limited to, wading with minimal threat to public health as a result of water quality.

Standards: **Fecal coliforms** - shall not exceed 5,000/100 mL in more than ten percent of the samples taken during any thirty - day period.

E. coli - shall not exceed 576/100 mL in more than ten percent of the samples taken during any thirty - day period.

This is an interesting interpretation of the federal criteria guidelines. First, what is called primary recreation in the federal documents is called "bathing waters" use in the state of Ohio Water Use Designation. It was stated in the preceding section that the federal criteria are clearly intended for actively bathing waters and not for accidental swimming.

Second, the use that is called "primary contact" and the appropriate standards is defined as "secondary use" in the USEPA (2000) guidance document and is very similar to the abolished secondary use in Illinois. The Ohio's "secondary use" is a state defined use for waters that do not meet the bathing waters and primary use designations. The more lenient "primary use" is designated to all impounded waters classified as modified warmwater use.

By allowing a choice between the E. coli and fecal coliform standards it is quite likely that the more lenient FC standards will determine the use.

selection of criteria for carcinogenic compounds. The USEPA (2002) guidance document states that a use attainability analysis as described in the federal regulations (40 CFR 131.40) *is not needed* for selecting the risk between 8 to 14 illnesses/1000 people and the risk selection is at the discretion of the state. Therefore, this UAA will only make a recommendation on the risk. Consequently, Illinois EPA and the Illinois Pollution Control Board can define more than one category of primary recreation. The US EPA (2000, 2002) guidelines also provide a rationale for seasonal water quality standards for states in northern climates. Again this option is up to the discretion of the state and does not require a UAA.

Previously, for example in the *Water Quality Standard Handbook (USEPA,* 1994), US EPA allowed consideration *of natural or background* bacterial contamination of animal origin. However, USEPA has changed this position after finding that gastrointestinal illnesses can occur after exposure to microorganisms of nonhuman origin, *Escherichia coli* being a widely publicized example. Other disease causing microorganisms such as *Giardia* and *Cryptosporidium,* are also frequently of non human origin, originating, for example, from cattle. Only if a significant portion of the fecal contamination is demonstratively caused by migrating waterfowl, resident wildlife population, or wildlife refuges and is potentially uncontrollable, and/or the primary recreation is not achievable by controlling other sources, the state may assign an intermittent wildlife impacted, or secondary use.

The USEPA guidance also allows high flow exemptions but only in cases where high flows prevent the primary contact recreational use. If the water body is impacted by combined sewer overflows, the supporting analysis should be consistent with a Long Term Control Plan. This means that, in the case of the Lower Des Plaines River, the long term plan of the TARP project, not just the present situation, should be considered in defining the risk and magnitude of the standard. The high velocity cutoff suggested in the guidance document is not applicable to the Des Plaines River because of the large hydraulic capacity of the channel. An example of high velocity restriction is a floodway canal.

The US EPA (2002) guidelines now clearly specify what should be basis for assigning a use other that primary use:

- Is the water body publicly identified, advertized, or otherwise regularly used or known to the public as a beach or swimming area where primary contact recreation activities are encouraged to occur?

- What is the existing water quality? If it is not currently meeting the applicable recreational water standards, do the exceedences occur on a seasonal basis, in response to rainfall events, or at other times due to other conditions or weather related events?

- Is the primary contact recreation use attainable through the application of the effluent limitations under CWA Sections 301(b)(1)(A) and (B) and 306 or through cost effective and reasonable best management practices for nonpoint sources?

- What are the sources of pollution within the waterbody? What are the relative contributions of these sources?

Summary on Modification of the Use in Non-Primary Contact Recreational Waters

- A primary recreation use should be adopted on any water body where people engage or are likely to engage in activities that could result in ingestion of the water or immersion such as swimming, kayaking, water skiing, or others.

- Special attention should be focused on the behavior of children that are more likely to engage in such activities even on water bodies where adults would not.

- States, through a UAA, may change the primary recreation use to another use such as
  intermittent
  secondary, or
  seasonal

- In some cases, recreational uses may be removed altogether such **as**
  - the primary recreation is not an existing use
  - waters that are irreversibly impacted by wet weather events
  - where climate allows primary contact recreation to occur only on a seasonal basis
  - meeting the primary recreation would result in wide spread adverse socio-economic impact
  - water access is prevented by fencing
  - an urban water body serves as a shipping lane

Physical factors alone would not be sufficient justification for removing or failing to designate a primary contact recreation use. In making the UAA decisions, the state should consider a combination of factors such as

- the actual use (is primary recreation an existing use?)
- existing water quality
- water quality potential
- access
- recreational facilities
- location
- safety considerations
- physical conditions of the water body

"Access" implies restricted access, meaning the water body is fenced off. Remoteness is not a valid basis for an attainability decision on recreation.

Selection of Secondary Contact Recreational Use

The criteria guidance documents do not provide guidance as to the protection of water bodies that are effluent dominated. One could make an assumption that, based on the USEPA (1986) bacterial criterion wording and magnitude of the standard, such water bodies would not be recommended for primary recreation, public beaches would not be present and, if recreation occurs, swimming would be incidental and discouraged by posting signs and other restrictions (railings or fencing).

The proposed rule in USEPA (1988) presented an example of the interpretation that appears to be pertinent to the UAA for the Lower Des Plaines River:

> Suppose a city has created a greenway along a stream that receives wastewater effluent upstream of the greenway and has posted "no swimming" signs. The greenway attracts children leading to inevitable "unauthorized" swimming. If the physical conditions of the stream are suitable for swimming, the swimming occurs on a frequent basis and the greenway provides recreational facilities and access, the only factor limiting the use may be a water quality problem that in the judgement of the state can be controlled to achieve the primary contact use. The linkage between existing and designated uses encourages an evaluation of the full suite of factors making a decision whether or not primary contact recreation should be protected.

This possible interpretation of the rule implies that if the only reason for the existing nonswimmable water body use is an upstream wastewater discharge, Reason 6 (widespread adverse economic impact) documented by the UAA would be the only possible allowed reason for removing the primary recreation use. However, the proposed rule advises that the state (i.e., UAA preparers) look at a suite of factors such as those listed above. Also in the proposed rule, the USEPA revealed that it was considering whether the regulation should be amended to allow consideration of the physical factors, as the basis for removing or not designating the primary contact use.

The proposed rule also, in some cases, pointed out that liability questions may lead the state to propose a secondary use but implement standards that would be commensurate to the primary use. The issue for the state would be to strike a balance between two concerns: (1) the possibility of inadvertently encouraging swimming where it should not occur because of safety considerations and (2) protecting that use if it did occur.

Where states adopt a use that is less than primary contact recreation, federal regulations require reexamination every three years to determine whether new information has become available that would lead to the designation of a more protective use.

The USEPA (1986) document provides the magnitude of criteria only for primary recreational use on frequently used beaches. No guidance is given for secondary use. The USEPA (1986) document gives only cursory attention to other water bodies that do not have public frequented beaches. It suggests that other recreational resources such as wading ponds used by children *or waters where incidental full body contact occurs because of water skiing or other similar activities* should also receive some protection.

The USEPA (2002) document somewhat clarified the issues of the application of the secondary use standard. While the quantitative magnitude of the standard is not included, the document states that *a secondary contact recreation use may be applicable to waters that are, for example, impacted by human caused conditions that cannot be remedied, or where meeting the criteria associated with the primary contact recreation use would result in substantial or widespread social and economic impact.*

For water bodies where it is demonstrated through a UAA that primary contact recreation will not occur, adoption of a recreation use and water quality criteria to protect secondary contact activities may be appropriate. The secondary use is defined by the USEPA (2000, 2002) as those activities where a low percentage of participants would have little direct contact with water and where ingestion of water is unlikely, such as wading, canoeing, motor boating, fishing, etc. The USEPA recommends developing a secondary contact criterion that would not exceed a geometric mean of five times EPA's recommended water quality criteria for primary recreation. However, it is not clear which risk should be taken as the base for the definition of the secondary standard. For example, if the risk of 8 illnesses/1000 is used, the secondary standard would be 5 x 126 = 630 EC cfu/100 mL, if 14 illnesses/1000 swimmers is used then the risk would be 5 x 548 = 2,740 EC cfu/100 mL. The risk to swimmers can be estimated from Figure 7.1 although the data for the line extrapolation were not provided for the risk greater than 14 gastrointestinal illnesses per 1000 swimmer.

Monitoring and Number of Samples to Define Existing Uses and Compliance with the Standard

For routine sampling of rivers (e.g., the Des Plaines River) that do not have highly frequented public beaches nor are used for water supply, data series containing five samples per month (or 30 day periods) are generally not available and the geometric mean criterion cannot be evaluated for a representative 30 day period. The guidance documents (USEPA, 2000, 2002) recommend that in such a case all available samples are evaluated, i.e., the geometric mean is calculated using all the samples. Thus, a scientific judgement based on the probability distribution and extrapolation will have to be used for the evaluation.

It is not clear whether just one 30 day period with more than 5 samples is sufficient or if sampling should be done continuously over the entire bathing season. One would presume that, because this criterion protects swimmers, sampling may not be necessary during nonswimming periods, e.g., winter, late fall and spring. This is confirmed in the narrative of the USEPA (1986) criteria document that describes the studies only for the swimming season. The criteria document also states that designated public beaches require the most rigorous monitoring, which is logical, and the standard was developed for such situations. Such areas are frequently lifeguard protected, provide parking and other public access and are heavily used by the public.

Interpretation of the General Use Standard and USEPA Criterion for Sites that do not have Sufficient Number of Samples

It was documented throughout this document that MWRDGC sites are sampled approximately weekly and IEPA sites nine times per year. This frequency is not sufficient to obtain exact compliance or noncompliance with the standards. In this document compliance and excursions will be interpreted as follows:

1. The geometric mean of all samples will be used to evaluate compliance with the lower standard of 200 FC cfu/100 mL. The geometric mean can be estimated either as an antilog of the mean of the logarithms of the FC cfu/100 mL measured values or as a 50 percentile

on the log - normal probabilistic plot. If the values of the measured FC concentrations followed exactly the log normal probability distribution, the two means would be identical. If the sample series contains outliers, the 50th percentile value is a more realistic value.

2. The higher Illinois standard of 400 FC cfu/100 mL will be compared to the 90 percentile of the measured fecal coliform densities. It is believed that if the 90th percentile complies with the standard, then any 30 day period would also comply.
3. The federal one sided confidence limit will be estimated using the USEPA category for infrequent use (95th percentile) and the logarithmic standard deviation obtained from the plotted and analyzed data series for the site. Table 7.1 presents the maxima calculated under the assumption that the standard logarithmic deviation of the collected data equals 0.4. This may not be the case for the Lower Des Plaines River. In this case, the US EPA (2002) guidelines allow recalculation of the single maximum limit using the logarithmic standard deviation based on the measured data. If the standard deviation is not 0.4 then the single maximal value limit can be calculated as

$$C_{\max} = 10^{[Log(geometric\ mean) + 1.65 x LogSD]}$$

where Log SD will be calculated or read as a difference of the 84th and 50th percentile values of the logarithms of the cfu/100 mL concentrations.

Relation of E. coli to Fecal and Total Coliform

The total coliform group is defined as those facultative anaerobic, gram-negative, non-spore forming, rod shaped bacteria that ferment lactose with gas formation within 48 hours at temperatures of 35°C, or, as applied to a membrane test methodology, produce a dark red colony with a metallic sheen within 24 hours on an Endo-type medium containing lactose (Clescieri et al., 1998). Total coliform counts may include bacteria that are of both fecal and nonfecal origin. Thus, the test for total coliforms is not conclusive and is not used today for assessment of fecal pollution or suitability of water bodies for contact recreation. Fecal coliforms, in a similar test, are grown lactose on or ferment lactose with gas production at an elevated temperature of 44.5°C.

Previously, total and fecal coliforms were used as indicators of bacterial pollution. Total coliform densities (that also include fecal coliforms) are much larger than those of fecal coliforms when measured on the same sample and may include organisms that have another origin, e.g. from soil. Typically, total coliform/fecal coliform ratios measured in the Ohio River by ORSANCO (1971) were about 7, and that for the Upper Illinois River measured by Butts, Evans and Lin (1975) were about 11, respectively. Both reports indicated a wide range of ratios.

Escherichia coli is a member of the fecal coliform group of bacteria and, consequently, a member of the indigenous fecal flora of warm-blooded animals. *E. coli* microorganisms are defined as bacteria giving a positive total coliform response and possessing an enzyme that releases fluiorogen that can be detected under ultraviolet light (Clescieri et al., 1998). For the Upper Illinois River, using data from the NAWQA program reported in Terrio (1995), the densities of fecal and *E.coli* densities were about the same (Figure 7.2). *E. coli* have been found to cause gastrointestinal diseases that sometimes were fatal. The data from the report by Terrio (1995) are the only data available that relates the densities of *E. coli* to fecal coliforms in the Upper Illinois River basin. Figure 7.2 shows that the ratio is about 1:1. However, this ration is unlikely because the *E.coli* is a subgroup of fecal coliforms. Other studies show that the ratio is less than that. Calderon *et al.* (1991) measured mean *E.coli* in a pond used for recreation as 51 EC cfu/100 mL (cfu = colonies forming units) and the density of the fecal coliforms was 62 FC cfu/100 mL. This would imply *EC/FC* ratio being 0.8 or 80%. Calderon et al. also found a very high correlation between *E.coli* and fecal coliforms (r^2 = 0.82). The Calderon *et al.* study was made on a recreational pond that had no point source sewage inputs and the source of fecal contamination was from wildlife. They identified the primary source of illnesses related to water contamination by pathogens the transmission of pathogens from swimmer to swimmer and not by the fecal pollution by pathogens from wildlife. In a French study on a river frequented by campers and recreationists and polluted by sewage, Ferley *et al.* (1989) found the *EC/FC* ratio of about 60%. This ratio would correspond to the magnitude of the 1986 USEPA criterion (126 EC cfu/100 mL) when related to the Illinois General Use Standard(200 FC cfu/100 mL), i.e., if the *EC/FC* ratio is 0.63, the criteria of 126 EC cfu/100 mL and 200 FC cfu/100 mL would be similar. However, if the *EC/FC* ratio is greater than 0.63 then the USEPA criterion

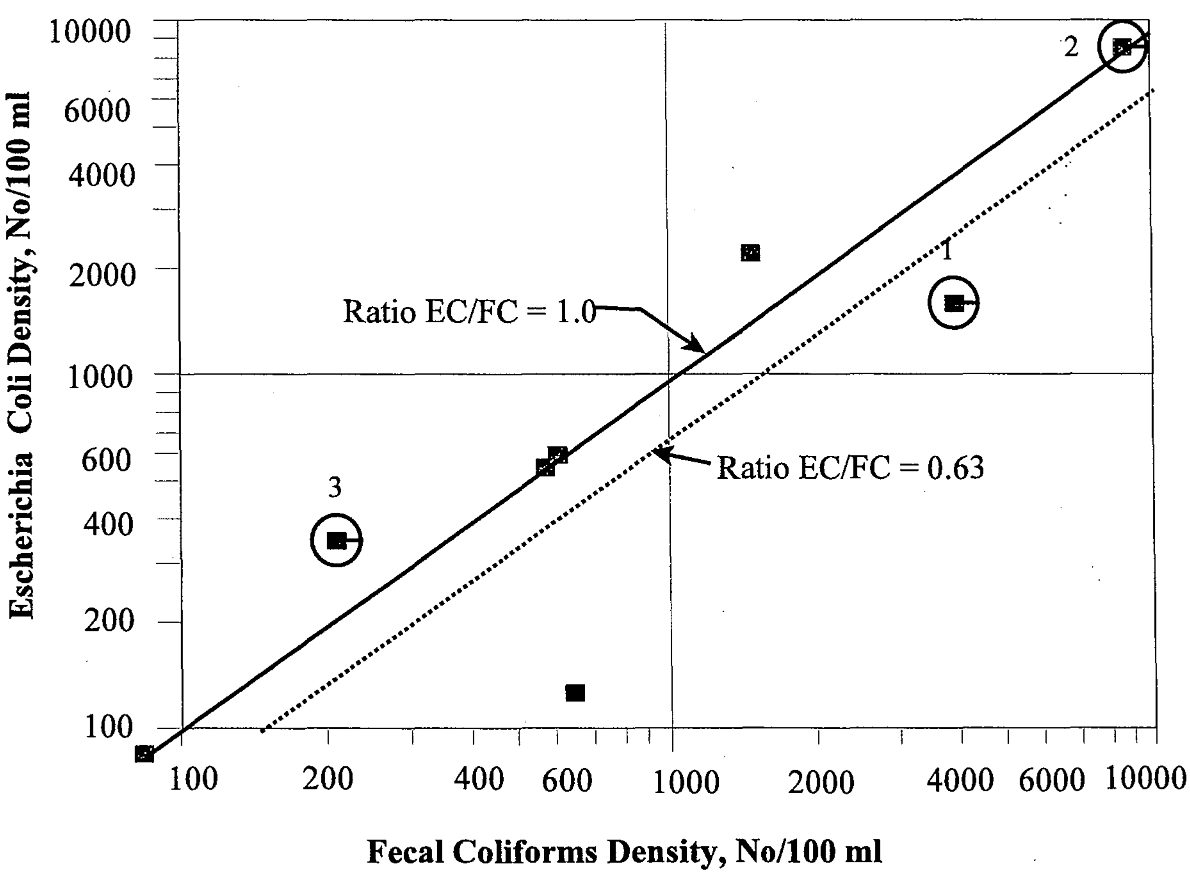

Figure 7.2 **Relation of escherichia coli to fecal coliforms densities for the Upper Illinois River (including the Des Plaines River). Data from Terrio (1995). 1 - Illinois River at Marseilles, 2 - Chicago Sanitary and Ship Canal, 3 - Des Plaines River in Riverside.**

based on *E.coli* is more stringent than the state standard of 200 FC cfu/100 mL. However, these standards, based on the latest USEPA (2002) guidelines, are only applicable for highly frequented beaches and not to the Des Plaines River. The scatter of data on Figure 7.2 is such that the statistical possibility of the EC/FC ratio of being 0.63 cannot be excluded. This ratio is shown on the figure as a dashed line.

It is clear, on one side, that the ratio of EC/FC is highly variable and cannot be used for regulative purposes, i.e., permits for fecal coliforms in the effluent cannot be directly related to the *E.coli* ambient standard by the ratio. On the other hand, **since the *E.Coli* is a part of the fecal coliform group, the ratio cannot be one or greater.** Thus, the information fecal coliform densities provides a good surrogate for analysis and judgement for implementation of the E.coli based standard.

The USEPA's 1986 criteria also suggests using enterococci as indicator organisms. The enterococcus group is a subgroup of the fecal streptococci. The difference between the definition of streptococci and enterococci is the ability of the enterococcus group to grow in high salinity water (Clescieri et al., 1998). Therefore, enterococci are recommended as indicator microorganisms for marine beaches. *The E. coli* group is included in the fecal coliform test and they are highly correlated with fecal coliforms, as documented in the previous paragraph. However, enterococci are not a part of the fecal coliform group. Nevertheless, in the Calderon *et al* (1991) study, densities of fecal coliforms, *E. coli*, and enterococci were significantly correlated with each other, i.e., as one increased in density the other two also increased. *E. coli* is the preferred indicator organism for fresh water swimming areas.

The bacterial densities in the Lower Des Plaines River are much higher than those measured in the above studies.

Water Body Assessment

History of the Standard

In the 1970s, the Illinois Pollution Control Board (IPCB) adopted two standards for the Upper Illinois River Waterway (IPCB, 1972, Butts et al., 1975) one for the general use and the other for the restricted use. The restricted use standard was approximately five times the general use standard. Based on this standard, treatment plants discharging into the CSSC and the Des Plaines River were chlorinating the effluents.

In the 1970s and early 1980s, adverse effects of chlorination and residual chlorine on the aquatic environment and public health was discussed extensively in the literature (Haas et al., 1988). Further more it was found that coliform bacteria may regrow after chlorination in the receiving water bodies and effluents (Shuval H., et al., 1973; Haas et al., 1988). It should be pointed out that all these adverse effects were related to residual chlorine in the effluent and receiving waters because the effluents were not dechlorinated. Current practices almost always require and implement dechlorination (sometimes with reaeration) after chlorination to mitigate the adverse effects of residual chlorine.

Because chlorination of effluents from treatment plants resulted in limited benefits to the receiving waters, and because of possible adverse effects to aquatic life and human health, the Illinois Pollution Control Board ruled in favor of stopping chlorination of effluents into secondary (restricted) use waters. Chlorination and disinfection of any type of effluents located on the secondary contact waters was discontinued in 1983 or 1984 (Terrio, 1994). The secondary contact waters include the Lower Des Plaines River from Lockport to the I-55 bridge, Chicago Sanitary and Ship Canal and Calumet Sag Channel. After the repeal of the numeric standard for secondary contact recreation in the secondary contact waters, utilities located on these waters stopped disinfecting the effluents. Disinfection continued on effluents located on the Des Plaines River upstream of Lockport.

Current and Historical Densities of Fecal Coliforms in the Lower Des Plaines River

Effect of Cessation of Chlorination on the Bacterial Densities

Chlorination was discontinued in the Stickney water reclamation plant in April 1984, at the North Side plant in March 1984, and at the Calumet plant in August 1983. In 1985 the TARP system was put into operation. That significantly reduced the number of CSOs into the Chicago waterways. The USGS (Terrio, 1994) analyzed the impact of discontinuing chlorination and concluded that the effect on the increase of the bacterial densities using fecal coliforms as indicators extended 6.8 miles downstream from the Stickney effluent discharge, which is upstream from the confluence of the CSSC with the Calumet Sag Channel. This negligible effect of discontinuation of chlorination of fecal coliform densities on the receiving waters further downstream was also confirmed by Haas et al., (1988) and can also be derived from the MWRDGC report by Sedita et al. (1977). The study by Haas at al. is limited to Calumet Sag Channel and Calumet WWTP (water reclamation plant). Sedita et al. and Torio's studies include data and analyses of the effects of cessation of chlorination at the three major plants discharging into the Chicago waterways (CSSC and Calumet Sag Channel).

This UAA is not focusing on the Chicago waterways; however, the Chicago Sanitary and Ship Canal is the main contributor of flows and pollutants to the Lower Des Plaines River. As stated in the preceding section, there are no data available in the Lower Des Plaines River on the densities of E.coli or enterococci indicator organisms. The collected samples have been analyzed for total and fecal coliforms in an old study by Butts et al.(1975). The NAWQA study by Terrio (1995) did analyze concurrent fecal and escherichia coli but no sampling was made in the investigated reach of the Lower Des Plaines River. The nearest NAWQA sampling locations were on the Des Plaines River at Riverside, Chicago Sanitary and Ship Canal at Romeoville, and Illinois River at Marseilles. This sampling provided information on the relation of the fecal coliforms vs. escherichia coli shown on Figure 7.2. As pointed out, studies indicate a good correlation between the *E. coli* and FC densities and the *E. coli* density should be less than the density of fecal coliforms.

Figures 7.3 and 7.4 show the probabilistic plots of fecal coliform densities obtained at the IEPA G-23 and MWRDGC monitoring sites located in the Brandon Pool and MWRDGC 94 and 95 sites in the Dresden Pool. The MWRDGC 95 sampling site is located at the I-55 bridge, RM 278. For historic comparative purposes the measured densities measured by Butts et al. (1975) were also plotted. Butts et al. measurements were made at a time long before TARP was built. Current measurements reflect the effect of TARP but no chlorination. The difference between the 1971 data and the current data is surprising and shows the tremendous beneficial impact the TARP and wastewater treatment projects of the MWRDGC and other actions taken upstream along the CSSC had on reduction of fecal coliform densities. 98 % and 96% reductions of FC densities was achieved in the Brandon and Dresden Island Pools, respectively between 1971 and 2000 in spite of cessation of chlorination in the1983-1984 period. However, the densities of fecal coliforms in the two pools are still above the general use standards of 200 FC cfu/100 mL for the geometric mean and less than 10% excursions of the maximum standard of 400 FC cfu/100 mL.

The probabilities of excursions of the Illinois General Use Standard (the probability of excursion in percent is 100 - probability of being less or equal) for the maximum allowed concentration (400cfu/100mL) and geometric means obtained from the probabilistic analysis of the IEPA and MWRDGC data are:

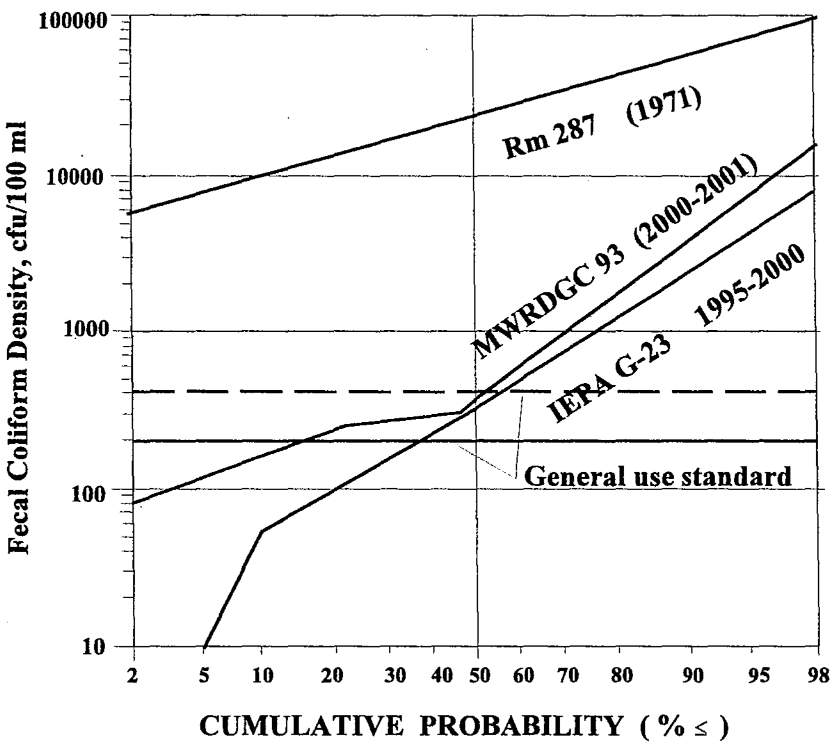

Figure 7.3 Densities of fecal coliform indicator organisms in the Brandon Pool. 1971 data from Butts et al. (1975)

effluents from the Stickney, Calumet and North Shore water reclamation plants represent a significant portion of the flow in the Des Plaines River in the two studied pools.

The selected reference water bodies were:

Kankakee River at Momence
Green River
Mackinaw River

Description of the reference water bodies and their watersheds are included in Chapter 2. None of these sites represent "pristine" conditions. However, the reference water bodies do not have major urban point sources of pollution and have relatively good riparian buffers in most of their length.

Figure 7.5 shows that the reference water bodies meet the Illinois General Use Standard of geometric mean of 200 FC cfu/100 mL when geometric mean (50 percentile) densities of fecal coliforms are considered. Because the maximum standard of 400 FC cfu/100 mL is exceeded with a probability of 25 to 45%, the Illinois maximum standard is not met. Based on the EC/FC ratios presented previously, it is likely that the federal criterion of the geometric mean of E.coli of 126 cfu/100 mL would be met in some reference water bodies. The single maximum standard of 408 EC cfu/100 mL (lightly used full body contact) or 576 cfu/100 mL (infrequently used full body contact) would not be met in these reference water bodies.

Conclusions on the Attainability of Standards in Reference Water Bodies

The probability distributions from the references streams were combined to yield a reference range represented by the shaded area on Figure 7.5. Figure 7.6 compares the current bacterial densities in the Dresden Island Pool with the reference conditions from Figure 7.5. Figure 7.6 shows that the geometric averages of the Dresden Island Pool fecal coliforms are about three times larger than those for the reference water bodies, in the probability range greater than 70% the densities (concentrations) would be about the same. This may be a common feature of less impacted streams in Illinois that, in general, are unable to meet the proposed federal criterion.

The probability of exceedence of the standard of the reference water bodies is as follows:

River	Probability of excursion of 400 FC cfu/100 mL (%)	Geometric mean cfu of FC/100 mL
Green River	40	205
MacKinaw River	29	140
Kankakee River	13	120

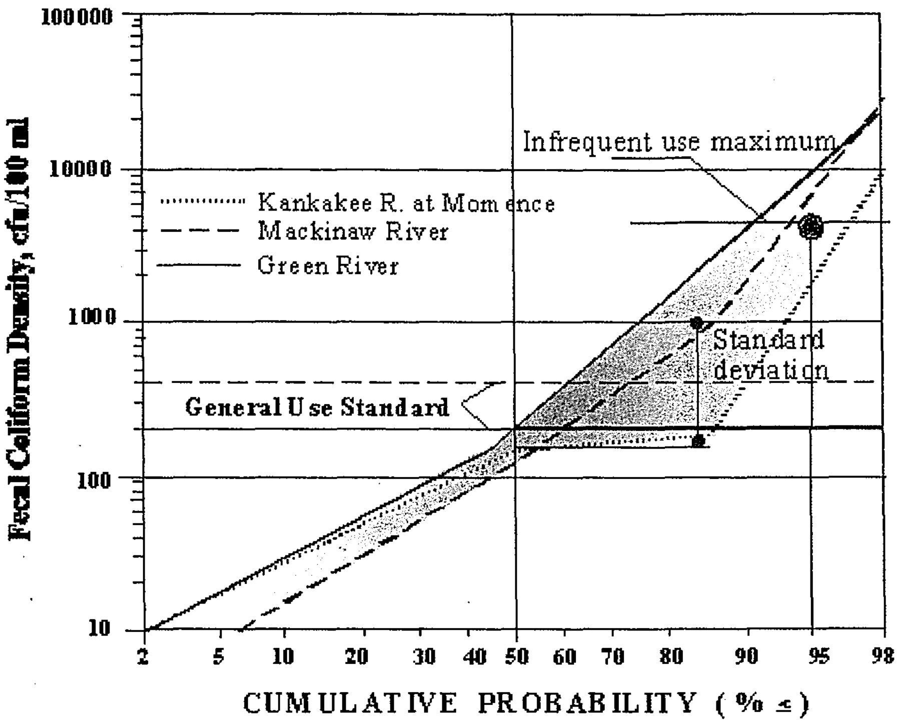

Figure 7.5 **Fecal coliform densities at reference water bodies**

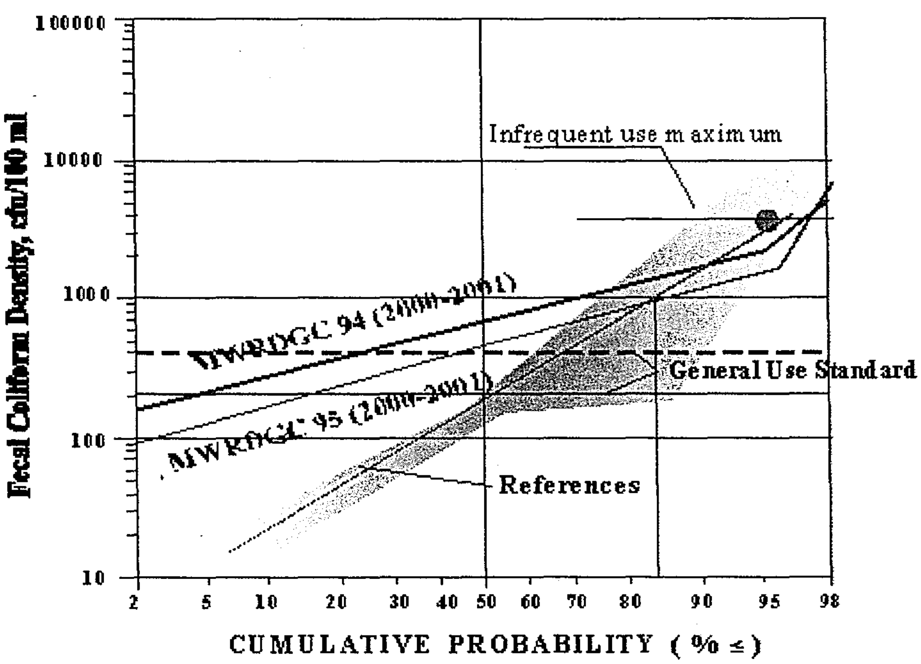

Figure 7.6 **Comparison of probability distributions of the reference fecal coliform densities with those measured in the Dresden Island (MWRDGC 94 and 95) Pool.**

Considering the fact that the reference unimpacted or least impacted water bodies do not meet the maximum Illinois General Use Standard for primary contact recreation, Reason 1 of the UAA regulation (CFR 131.10(g)) could be invoked. However, this approach is discouraged by the current USEPA (2002) draft guidelines. The same guidelines now allow the state to assign (without an UAA) a risk greater than 8 illnesses/1000 swimmers, up to 14 illnesses/1000 swimmers. Even with the greater risk (up to 14/1000) the single maximum, based on scientific judgement, is difficult to meet (may not be attainable) in these reference waters. However, the single maximum was calculated by the USEPA using the logarithmic standard deviation of 0.4. The approximate logarithmic standard deviation for the reference streams is larger, about 0.7. In this case, the 95% infrequent use single value maximum would be larger and the state could recalculate the values of these maximal values of the standard. However, because this concept is applicable to the *E.coli* and not to fecal coliforms indicator microorganisms, such recalculation would make sense only when adequate number of *E.coli* measurements on the Des Plaines River become available.

Features of the Lower Des Plaines River Impeding the Primary Recreational Use

Physical Limitation of the Pools for Primary Contact Recreation Use

It is the scientific judgement of the AquaNova/Hey Associates team that, based on the irreversible physical features, the use of the Lower Des Plaines River for primary recreation is limited (Dresden Island Pool) to almost impossible (Brandon Pool).

Brandon Pool (RM 291 to 286)

This pool of the Lower Des Plaines River, extending from the Lockport Lock to the Brandon Road Dam is essentially a constricted human-made navigation canal surrounded by the City of Joliet. Figures 7.7 and 7.8 show that the banks are vertical, made of concrete or sheet pile embankments. Riparian lands are highly developed, containing also the downtown of the City of Joliet. The embankments have two purposes, (1) to restrict the channel and allow urban development, and (2) protect the City of Joliet from flooding because the elevation of the downtown is below the water surface elevation in the Brandon Pool. Fencing or railings restrict and prevent public access to the river. The cross-section on Figure 7.9 documents that the channel is about 15 feet deep with vertical banks and no shallow (wading) areas almost in its entire length. Wading may be possible only in the Des Plaines River before the confluence of the river with the section of the Chicago Ship and Sanitation Canal (Illinois general use). The barge traffic is frequent with an average frequency of 8-10 barge tows per day and multiple barges towed. Based on the survey by AquaNova International, Ltd. (see the subsequent section), swimming was not observed and, because of the density of navigation and type of the channel, swimming should not be allowed for safety reasons.

The City of Joliet has developed a 10 acre park along the west side of the waterway in the Joliet City Center (Figure 7.7). Across from the park on the east side of the river, is the city's downtown. In addition, the city built a Riverwalk Promenade. The cultural park contains a theater/bandshell and picnic facilities but, currently, provides no access to the river itself. The park is purely for picnicking and visual observations/enjoyment of the river. Swimming, if it occurs, would be incidental and could be lethal to those who are not good swimmers, especially children. However, the City of Joliet has applied for a permit and state grants for a boat landing on the Brandon Road Pool north of Ruby Street (personal communication of Don Fisher, Joliet City Planning Department). Another proposed boating facility is being considered just north of Jackson Street. The facility will include townhouses with attached boat slips.

Figure 7.9 shows that a common multiple barge tow with a draft of 9 ft takes up a significant portion of the cross-section of the Brandon Pool. If two tows going in opposite directions meet, almost the entire cross- section would be taken up by the barges. This makes the Brandon Pool unsuitable for wide spread water borne recreation. Widening the channel and developing shallow areas for wading and swimming would require massive land acquisition in Joliet, relocation of the city center and astronomical investments that certainly would generate a wide spread socio-economic impact.

The physical attributes and the restricted use of the river in the City of Joliet are common to many urban streams throughout the world. In Ohio, such streams were included in a special use category called *"modified warm water use"* that retains most attributes of the general use but recognizes the fact that impounded waters cannot be compared to free flowing wadeable streams.

Dresden Island Pool (RM 286 to 277.8)

The investigated Dresden Island Pool extends from the Brandon Road Dam at the RM 286 to the I-55 bridge at RM 277.8. The pool is much wider (600 - 1300 ft) and not constricted by embankments. The pool, created by impounding the Des Plaines River for navigation, has established bank habitats with the dredged navigational channel in the center. Morphologically and qualitatively the pool can be divided into a three mile upper section of the investigated reach between the RM 286 and 283 and a lower section between the RM 283 and I-55 bridge. The upper section is not as wide (average width about 750 ft) as the lower section. Figures 7.10 to 7.13 show reaches of the Dresden Island Pool between the Brandon Dam and I-55. Figures 7.14 and 7.15 show the typical cross section in the Dresden Island Pool.

The upper section of the Dresden Pool surroundings between RM 286 and 283 are developed (Figures 7.10 and 7.13. Two power plants operated by the Midwest Generation are located in the reach. The lower section between RM 283 and the I-55 bridge at RM 277.9 is more natural with riparian habitat, oxbow lakes and wetlands surrounding the river. Several large chemical and other industries and a casino border the river. There are four marinas located on the Dresden Island Pool; however, none of them located upstream of the I-55 bridge. The nearest marina is just downstream of the I-55 bridge. Figures 7.11 and 7.12 and cross-sections of the Lower Dresden Pool show that waterborne recreation is possible.

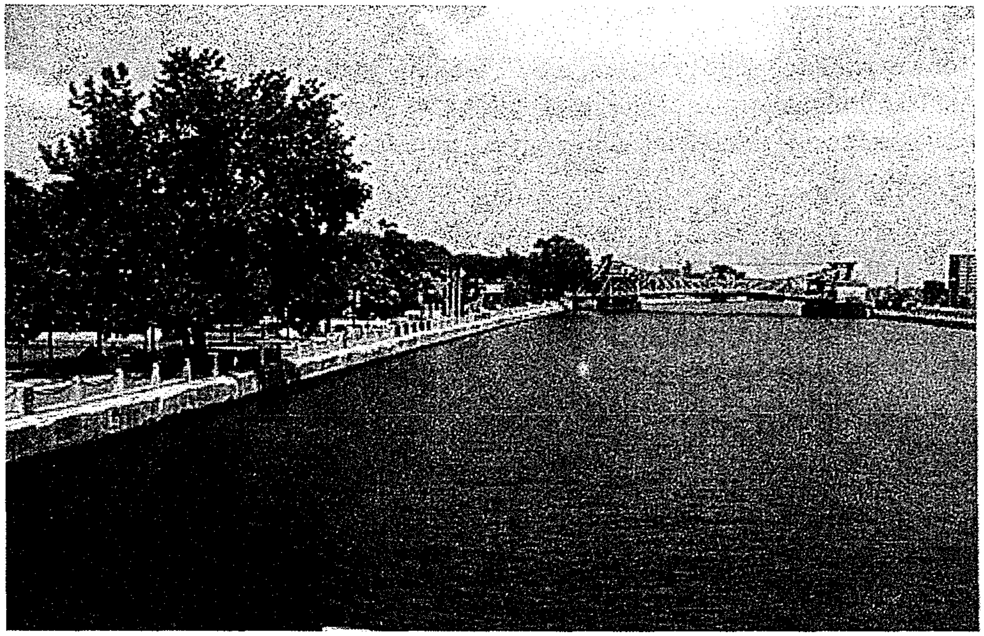

Figure 7.7 Lower Des Plaines River and Brandon Pool in downtown Joliet showing a narrow and deep channel with vertical embankments. Bicentennial Park is on the left side of the picture. The embankments prevent swimmers to climb back and the railing prevents access.

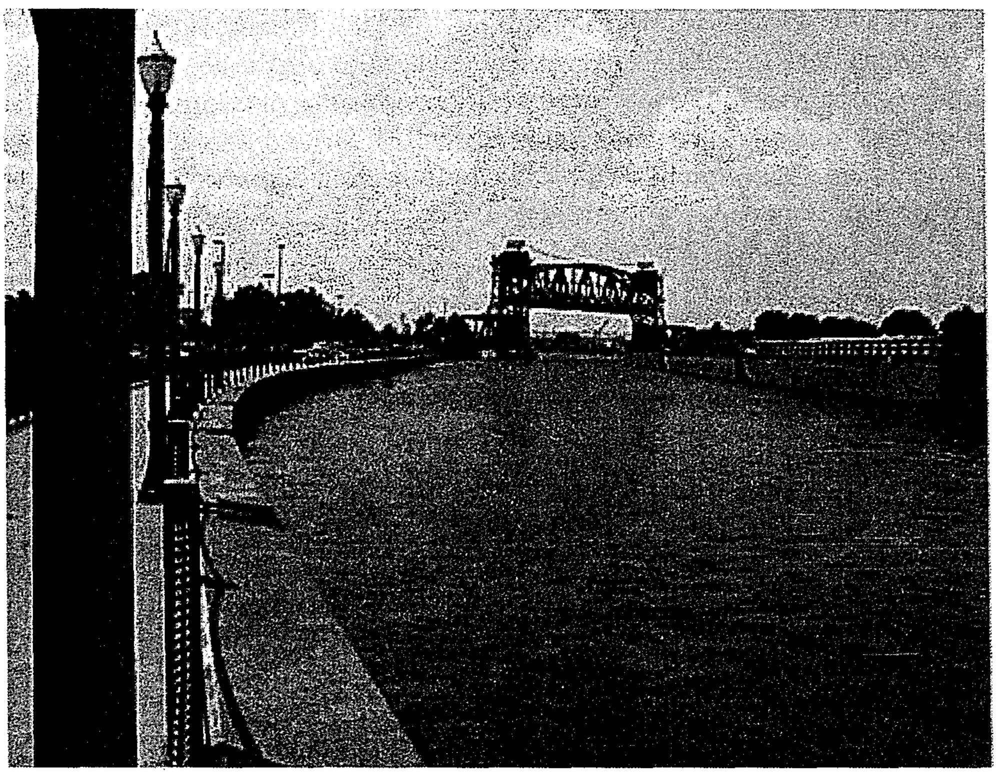

Figure 7.8 Brandon Pool of the Lower Des Plaines River in Joliet. Note vertical embankments and railing/fencing preventing access to the river.

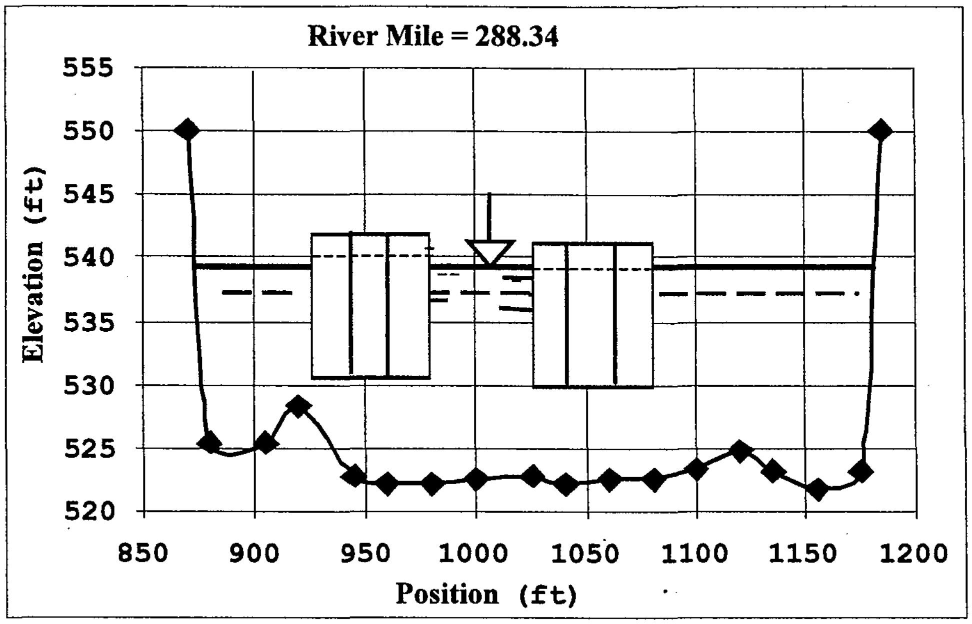

Figure 7.9 **Cross section of the Brando Road Dam pool with two barge tags indicating irreversible space limitations for recreation and vertical walls of the channel**

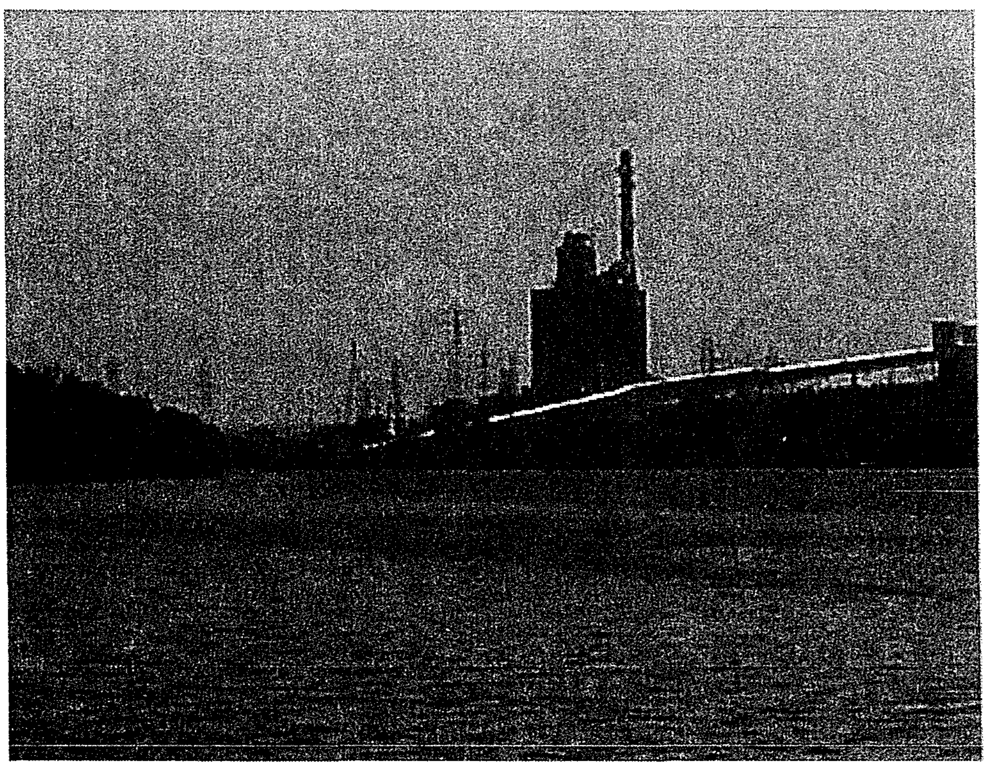

Figure 7.10 **Upper Dresden Island Pool near the power plants**

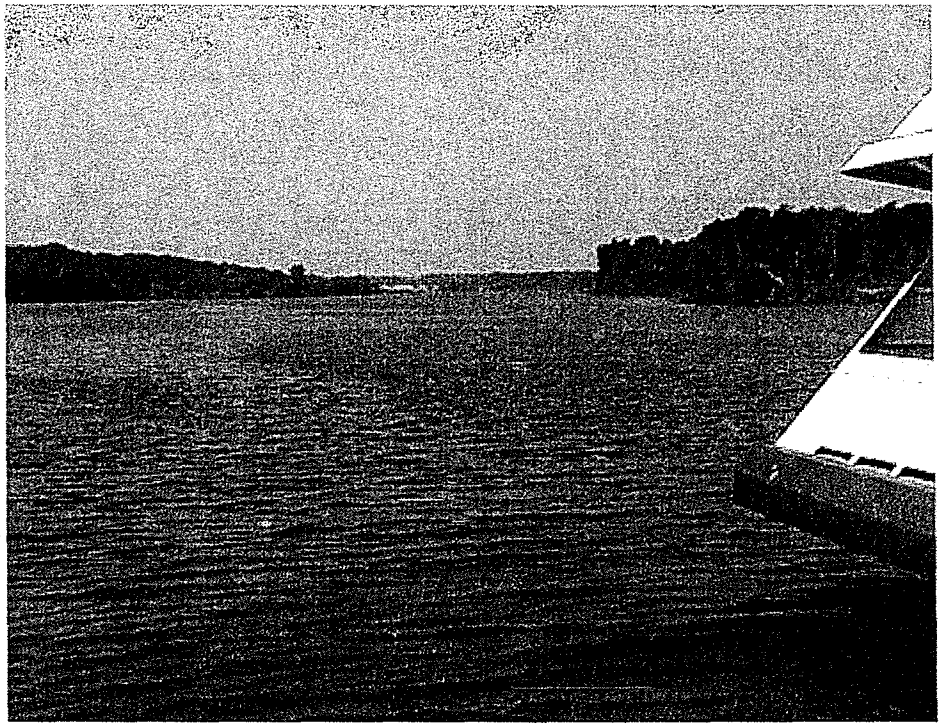

Figure 7.11 Lower Dresden Island Pool near Empress Casino.

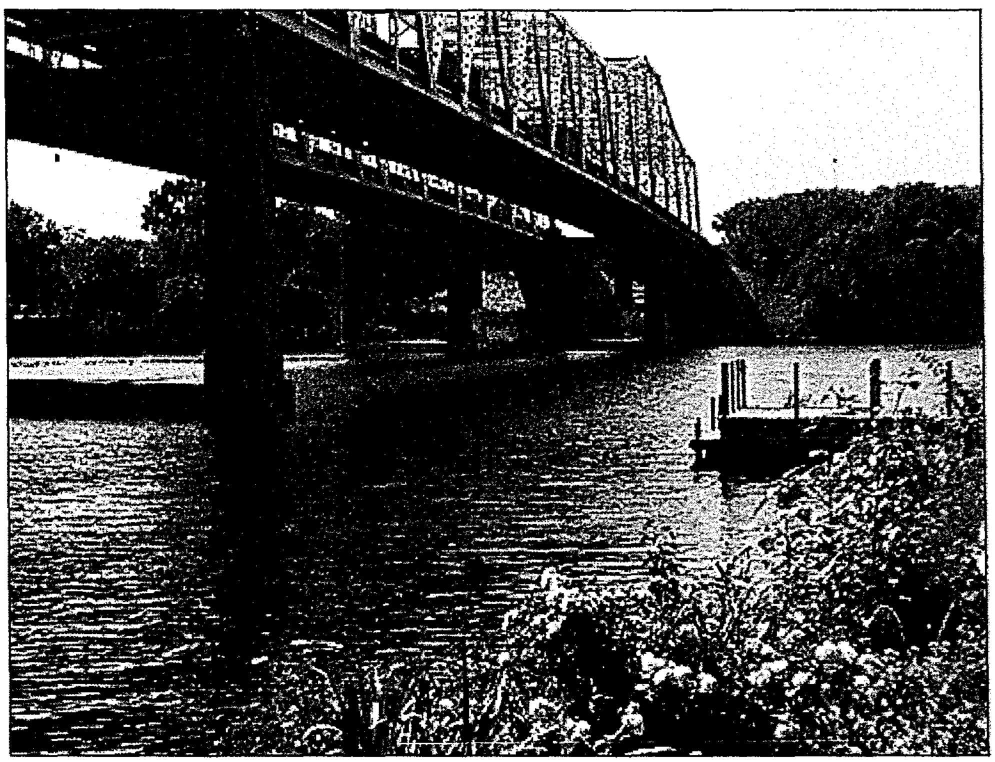

Figure 7.12 I-55 bridge on the Lower Dresden Island Pool - end of the investigated reach

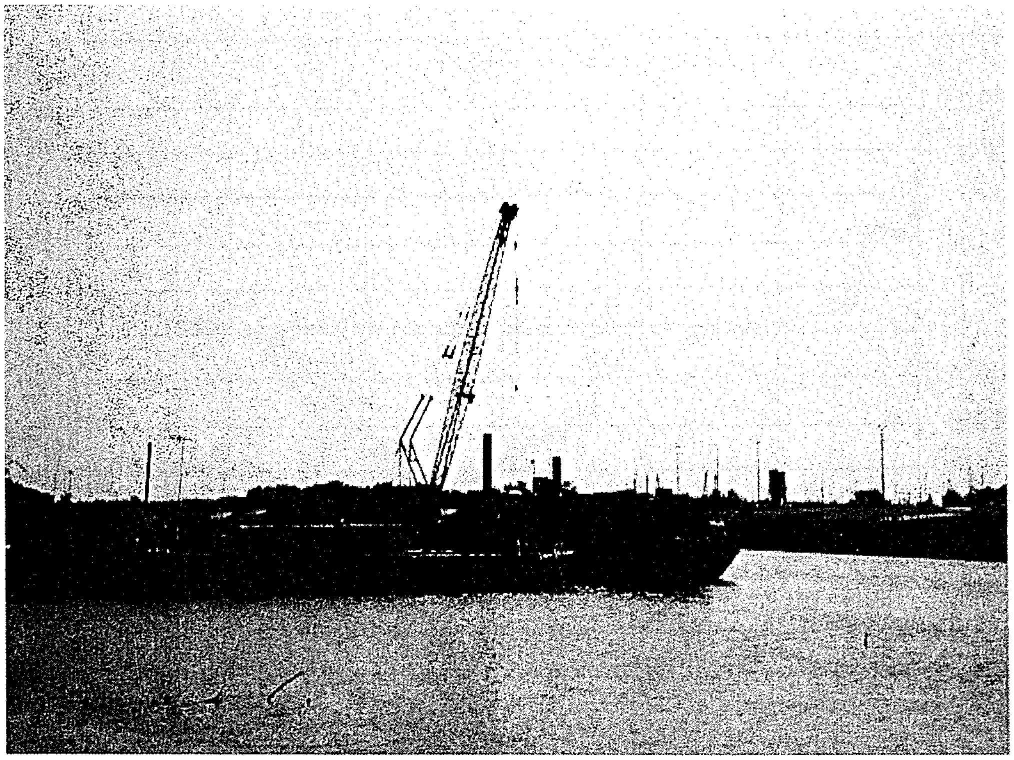

Figure 7.13 Upper Dresden Pool has some sections that are heavily used for navigation and industrial activities

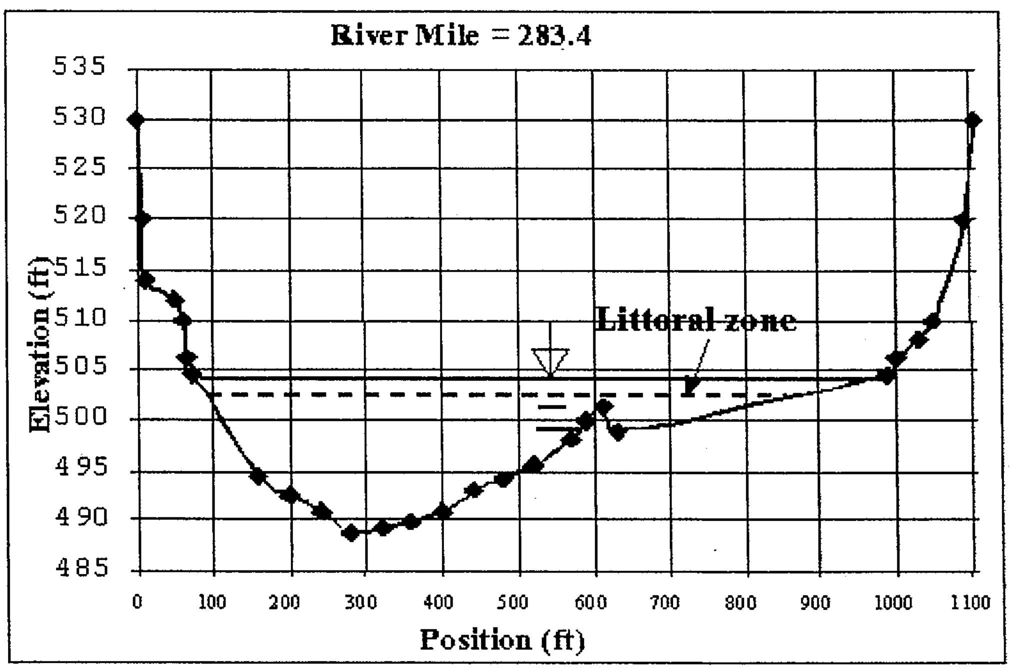

Figure 7.14 A cross-section in the upper section of the Dresden Island Pool

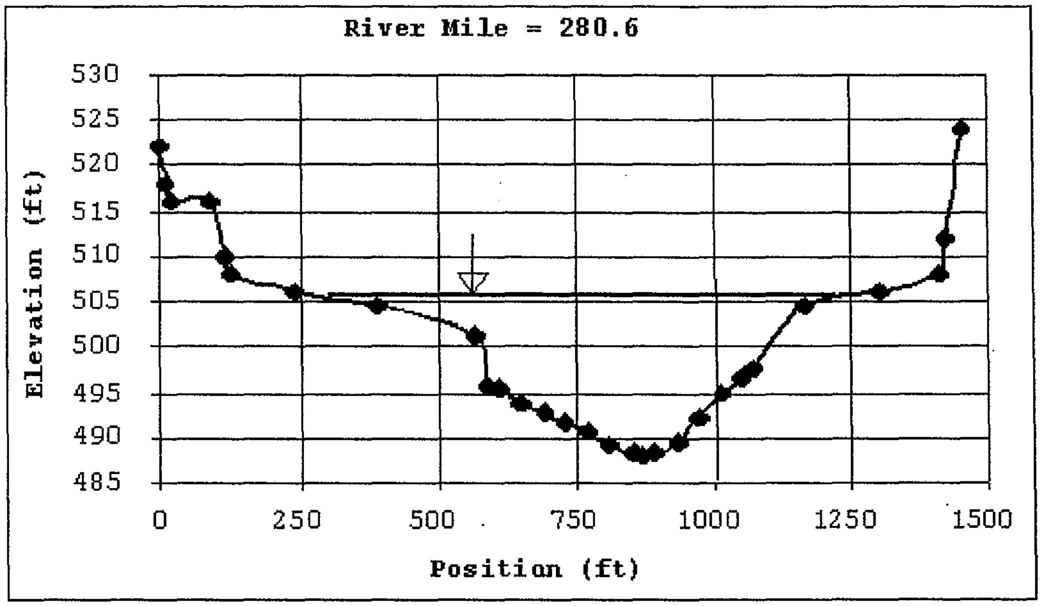

Figure 7.15 A cross-section in the lower section of the Dresden Pool Island pool upstream of I-55 bridge

Effects of Effluent Domination of River Flow and Urban Runoff on Primary Recreation

Point Sources

The second issue that must be addressed in order to assess the recreation potential is the fact that the river is effluent dominated and more than 90 % of flow is constituted by treated sewage effluents, combined sewer overflows and urban runoff. The design capacities of the North Side, Stickney and Calumet water reclamation plants (Terrio, 1994) and Joliet WWTP presented in Table 7.2. The location of the plants is shown on Figure 7.16.

The total effluent design flow from the two MWRDGC water reclamation plants represents 92% of the low flow in the Chicago Sanitary and Ship Canal that is then only marginally diluted by the low flow from the Des Plaines River downstream of the Lockport Lock. During average summer low flows the portion of the effluent flow in the Des Plaines River still may be more than 60%. The total average flow from all upstream public sewage treatment plants is 1870 cfs (see Table 1.1).

The distance of the Stickney plant from the confluence of the CSSC with the Des Plaines River is about 25 miles and that for Calumet discharge is almost 30 miles (Figure 7.16). With an average flow velocity in the CSSC of approximately 0.5 fps (less in the Calumet Canal) the residence time of wastewater in the canal to the confluence with the Des Plaines River and Brandon Pool is about 3 days, less during higher flows (e.g., wet weather flows with CSOs).

Butts, Evans and Lin (1975) developed a simple model for decay of fecal coliforms in the Upper Illinois River. The model is known as Chick's law and is expressed by the formula

$$\frac{N}{N_0} = 10^{-kt}$$

where N_0 and N are bacterial densities at time 0 and t days, respectively, and k is the die-off or death rate for the Upper Illinois River measured during warmer months (July to September) as being around 0.65day^{-1}.

Table 7.2 Typical flow magnitudes of major effluents and low flow in the receiving water bodies in cfs

		WWRP Flow* Design cfs	Average cfs	River flow 7Q10** cfs	Receiving water Body
North Side MWRDGC		514	367		North Shore Channel
Calumet MWRDGC	(RM 321)	546	290	350	Calumet River
Stickney MWRDGC	(RM 316)	1854	1007	1755	CSSC.
Joliet STP East and West	(RM 286 & 281)	44	21	1962	Des Plaines R.
Total			1685	1962	

* WWRP - Wastewater reclamation plant

** The river flows listed in the table are downstream from the effluents

By modeling the bacterial decay, using the fecal coliforms death rate of 0.65 day $^{-1}$ for the Upper Illinois River taken from Butts et al. (1975), the FC density (concentrations) could be reduced in three days by 99%, from the point of discharge at Stickney or Calumet to the Lockport Lock and dam. This would confirm Terrio's (1994), Haas et al. (1988), and Sedita et al. (1987) findings that the effect of discontinuing disinfection at the MWRDGC reclamation plants was limited to the CSSC.

Haas et al. reported the geometric mean of fecal coliforms densities in the Calumet water reclamation plant as 3,700 cfu/100 mL before 1983 (with chlorination) and 6,800 cfu/100 mL after cessation of chlorination. The geometric mean of fecal coliforms reported by Terrio for the Stickney plant was about the same with chlorination (3800 cfu/100 mL) but higher for the period without chlorination (19,000 cfu/100 mL). Reducing these concentrations by 99% will yield fecal coliforms density in the Lower Des Plaines River of 68 cfu/100 mL without and 37 cfu/100 mL before cessation of chlorination. These are indeed low numbers that would indicate a small effect of MWRDGC discharges on the fecal coliforms densities in Brandon Road and Dresden island pool. The sensitivity of the test is not such that it could detect the difference and these concentrations would be below the general use standard.

This does not imply that the MWRDGC plants do not have any effect on the bacterial densities in the Lower Des Plaines River. It only means that the difference between the bacterial densities before and after 1983-1984 may not be statistically distinguishable in the Lower Des Plaines River and the residual densities are small. Figure 7.17 shows the effect of ending chlorination on the densities of the fecal coliforms in the effluent from the Stickney WWTP and in the CSSC 11.7 miles downstream measured by Terrio (1994). Concentrations of fecal coliforms in the effluent increased by about an

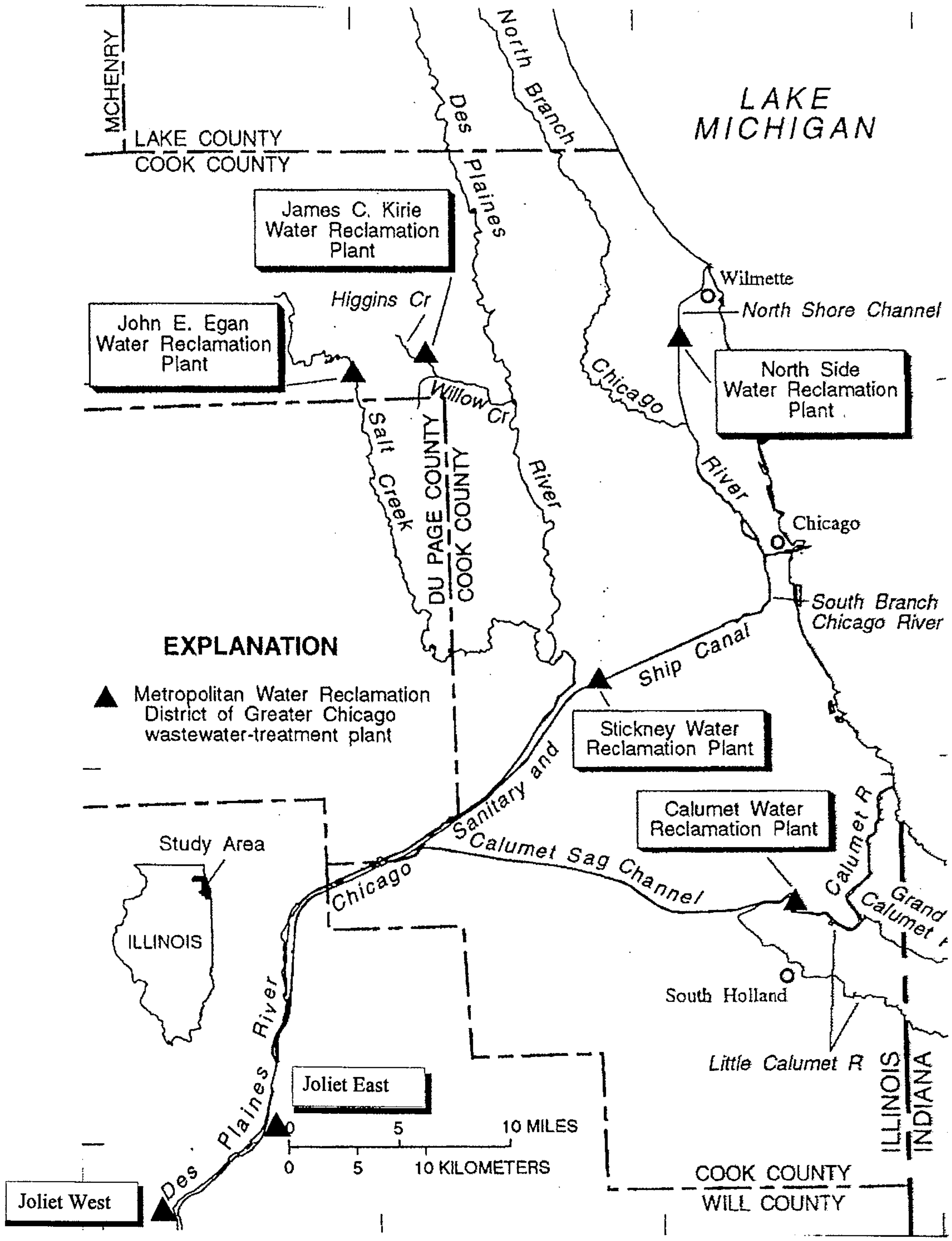

Figure 7.16 Location of the major wastewater treatment plant on the Des Plaines River and Chicago Sanitary and Ship Canal

order of magnitude but the effect 11.7 miles downstream was small and almost nil in the higher percentile (greater or equal to 90%) range. It appears that the high percentile concentration may occur during times of overflows from sewer systems in the Chicago metropolitan area. The geometric means (50th percentile) are significantly different, the period without chlorination showing 50th percentile densities about 66% greater then the period with chlorination.

The much smaller Joliet municipal wastewater treatment plants discharge into the Dresden Island Pool. The larger east plant discharges into Hickory Creek, near the confluence of the creek with the Dresden Pool just below the Brandon Road Dam at RM 286. A smaller Joliet West plant discharges its effluent directly into the Dresden Island Pool at RM 281. The effluents from these plants are not disinfected. Absence of disinfection in the Joliet plants and overflows from the sewer system in Joliet may have a greater impact on the Dresden Island Pool than those of the MWRDGC water reclamation plant because of less detention and decay of coliforms in the pool would be expected. It was documented in the preceding section that the Joliet effluent and CSOs increase FC densities in the Dresden Island Pool. It was pointed out that the City of Joliet is now completing sewer separation at the East Plant and the last CSOs should be eliminated by the end of 2006.

With the dilution ratio of the river flow vs. the Joliet effluent flow being about 100:1, the fecal coliforms density increase of the geometric mean in the Dresden Island Pool, assuming the Joliet effluent concentration of fecal coliforms of 19,000 cfu/100 mL (similar to Stickney and Calumet plants), could be as high 200 cfu/100mL which is not far from the measured difference of geometric means for Brandon and Upper Dresden pools shown on Figures 7.3 and 7.4.

The densities of the fecal coliforms in the treated effluents of the MWRDGC plants (before 1984) reported by Torio (1994) or Haas et al., (1988) and plotted on Figure 7.17 are much larger than those typical of the chlorinated/dechlorinated effluents today. While the geometric means for the Calumet and Stickney Plants with chlorination prior to 1983 were around 3000 - 4000 cfu/100 mL, current disinfection technology can achieve an order of magnitude smaller densities of coliform bacteria in the effluents. Typical current densities of fecal coliforms in disinfected effluents would have geometric mean less than 200 cfu/100 mL with a maximum of less than 4000 cfu/100 mL.

Effect of Combined Sewer Overflows

In the past, the river was severely impacted by wet weather combined sewer and storm sewer flows from Chicago and Joliet. Without TARP, CSOs of untreated sewage and wastewater were frequent and occurred about 60 times in an average year. The overflow numbers should be understood as one system event and not as a number individual overflows counting every CSO outlet.

Combined sewer overflows (CSO) from Cook County have been significantly reduced by the construction of the Tunnel and Reservoir Project (TARP) that intercepts the CSOs in the tunnel storage and significantly reduces the frequency of overflows to about ten to twenty per year. Additional storage and further reduction of frequency of overflows will be achieved by the construction of the reservoirs that will provide additional storage, now scheduled for completion by 2014, that, in relative probabilistic terms, may significantly further reduce the number of overflows

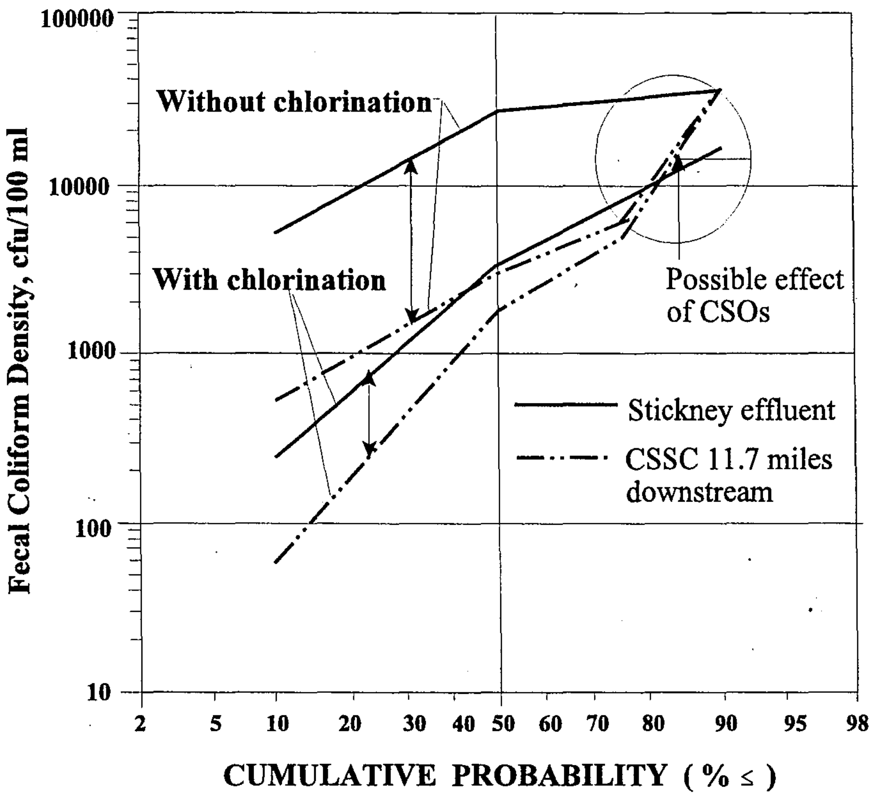

Figure 7.17 Probability of densities of fecal coliforms in the effluent of the Stickney treatment plant before and after cessation of chlorination and at a point on the CSSC 11.7 miles downstream (at the confluence with the Great Calumet River). Data from Terrio (1994)

to very low frequency, although the exact target frequency of overflows, per information by the MWRDGC, is not known. At the same time, the City of Joliet is separating the sewers and the last CSO point should be eliminated by the end of 2006.

The dramatic decrease of CSOs will result in significant reductions of the "high end" densities of fecal coliforms that will affect those that currently interfere with the current maximum exceedance standard and also would interfere with the single maximum standard for *E.coli* based on the current USEPA (2002) guidelines. The effect of the planned CSO controls (additional storage of TARP and sewer separation in Joliet) on the geometric mean standard will not be significant. The geometric mean concentrations can be lowered only by disinfecting the effluents.

Effect of Urban Runoff

The National Urban Runoff Project (USEPA, 1983) has measured at many sites throughout the US the concentration of pollutants in urban runoff. The results documented that urban runoff is polluted and controls are required.

The summary of the NURP studies are shown in Table 7.3.

Table 7.3 Fecal Coliforms Concentrations in Urban Runoff from NURP Studies (USEPA, 1983)

Warm Weather				Cold Weather			
No of observations		EMC cfu/100 mL	Coefficient of variation	Number of observations		EMC cfu/100 mL	Coefficient of variation
76	Median	21,000	0.8	52	Median	1,000	0.7
	Range	5,000-281,000			Range	350 - 330,000	

These relatively high densities of fecal coliforms in urban runoff are mostly of nonhuman origin. The results from the NURP sites consistently showed large seasonal differences between warm and cold months. Coliform concentrations during warm weather were approximately 20 times greater than those that occurred during colder periods. These differences were unrelated to comparable variations in human activities during these seasons.

High densities of coliform organisms were also observed in the Des Plaines River upstream of the confluence with the Chicago Sanitary and Ship Canal. (Table 7.4). Fecal coliforms densities at these stations should reflect mostly pollution by urban runoff and residual pollution due to chlorinated effluents of several smaller and medium treatment plants (see Table 1.1).

Table 7.4 Fecal Coliforms Densities in Des Plaines River upstream of CSSC

Location	Geometric mean Cfu/100 mL	Logarithmic standard deviation	95% high value
I-EPA G-11 (Lockport)	331	0.69	7,585
MWRDGC 91 (Lockport)	295	0.62	2,399
USGS Riverside (G-39)	1,905	0.51	19,952

A surprising fact is evident from Table 7.4, fecal coliforms densities at Lockport in the effluent dominated flow but with minimum urban runoff and some CSOs are much less than the densities measured in the upstream Des Plaines River that receives disinfected discharges and a large proportion of urban runoff. Thus, the necessity of dealing with the high bacterial contamination of urban runoff by implementing effective best management practices must be emphasized. However, the NURP study pointed out that although high levels of indicator organisms were found in urban runoff, the analysis as well as current literature suggests that fecal coliforms indicators may not be useful in identifying heath risks from runoff pollution and more E-coli data on urban runoff must be collected and analyzed.

Conclusions

The Lower Des Plaines River is effluent dominated and was also a CSO dominated water body. The fecal coliforms bacteria originate from multiple point and nonpoint sources. Reducing bacteria densities may require both disinfection of point sources (those that do not practice it today) and best management practices for nonpoint sources. The effect of point source effluents on bacteria density diminishes with the distance of the source from the Lower Des Plaines River. Therefore, the nearest sources to the river, the effluents from Joliet East and West plants that discharge directly into the Dresden Island pool, have a larger impact than effluent discharges from more distant MWRDGC plants on the Chicago waterways. Two studies commissioned by the MWRDGC and one independent study documented that the effect of disinfection at the MWRDGC plants on the bacterial densities in the Lower Des Plaines River would not be great and would be limited mostly to the CSSC.

Control of bacterial sources from diffuse urban runoff is difficult. There are no known places of water fowl or wild animal concentrations and the major diffuse source is urban runoff. The USEPA (2002) guidelines suggest that if some sources are uncontrollable more control may be required of controllable sources.

Return to disinfection would make sense if contact recreation becomes the designated use. This was recognized by the Illinois Pollution Control Board thirty years ago in its March 7, 1972 ruling: "*Summer disinfection of bacterially contaminated effluents ... has been required by the regulations for some time, with varying compliance dates and with more stringent requirements for discharges to waters designated for primary contact (swimming). ....the lower level prescribed for primary contacts should be readily achievable wherever disinfection is practiced. The additional safety seems well worth the additional cost in chemicals*". Disinfection is commonly required at WWTP's throughout the US and was practiced before 1983-1984 by the Metropolitan Water Reclamation District of Greater Chicago and is being practiced by all WWTPs located on the primary contact use waters, including the entire middle and upper Des Plaines River. It is evident that, due to the immediate proximity to the river, implementing disinfection in Joliet and other plants located on Hickory Creek will have a greater impact on the densities of the bacteria in the River than that at other more distant source. The type of disinfection would have to be carefully investigated because of the adverse effects of chlorine residuals on aquatic biota and public health. Today, a majority of treatment plants use chlorination with a follow-up dechlorination or non-chloride disinfecting methods. Fecal coliform densities in disinfected effluents are typically much less than 400 FC cfu/100 mL. Because disinfection was practiced before 1984 and, today, is a common part of most municipal treatment plant unit processes, implementing disinfection in WWTPs would not appear to constitute a widespread adverse socio-economic impact.

It should be pointed out that the effect of chlorination of MWRDGC would be small on Brandon Pool and almost negligible on the Dresden Pool. After completions of this UAA MWRDGC plants located on Chicago waterways would still be discharging into Secondary Contact waters without a bacterial standard. Need for disinfection of the MWRDGC located on the Chicago River and waterways will be contingent on the development of standards for the upstream (of Lockport) CSSC and the Chicago and Calumet rivers and Calumet channel. This UAA has documented a need for disinfection of the Joliet West and East plants' effluents and should extent to any municipal wastewater facility that has the potential to adversely impact the Des Plaines River.

Conflict Between the Navigation and Recreational Use of the Lower Des Plaines River

Navigation adversely affects the river recreation (Committee to Review the Upper Mississippi River - Illinois River Waterway, 2001; Becker, 1981; Graman et al., 1984). Recreational boaters respond to increased traffic by foregoing recreational boating and using their boats elsewhere. In the survey by AquaNova International (see the subsequent section) it was revealed that the waiting time at the locks for recreational boaters was up to four hours. The waiting time and the restriction on recreational boating during times of chemical cargo transportation were perceived as restricting the recreational use of the water body. The navigation frequency of barges and recreational boats in the Upper Illinois/CSSC Waterway, presented in Table 7.5, was provided by the US Army Corps of Engineers.

The Lower Des Plaines River is a part of the major US Inland Waterways. It connects the Chicago metropolitan commercial area and the Great Lakes with the Mississippi River and Gulf of Mexico. The value of the cargo shipped in Illinois is valued between \$ 2 billion and \$ 10 billion annually. The US Army Corps of Engineers, operator of the system, is planning modernization of the Illinois Waterway System to accommodate larger barge tows, up to 15 barges per tow (US Army Corps of Engineers, 2002). However, this planning effort has been the subject of critiques by environmental groups and by a panel of the National Academy of Science (Committee to Review the Upper Mississippi River - Illinois Waterway Navigation System, 2001).

The lock master at the Brandon Road Lock and Dam (Robert Smolka, personal communication) has stated that recreational boats are allowed throughout the locks, however with some restrictions. Barge traffic has priority over recreational boats and the recreational boats are not allowed in the lock with barges without the permission of the barge operator. Most barge operators do not allow recreational boats to go through with them for insurance liability reasons. If recreational boats are stacking up, every third operation of the lock is for these boats. Jet skies are not allowed in the lock unless they are tethered to another boat and the jet ski operator is out of the water.

According to the US Army Corps of Engineers regulations there is a restricted zone 500 ft above and 250 ft below the federal dams where boats are not allowed (personal communications of Jim Stimen, Rock Island USACOE District). Generally, these zones are marked with navigational buoys. The Coast Guard is responsible for enforcement of the federal boating regulations and adherence to buoy restrictions. The Coast Guard has no special regulations for boat activity near locks and dams.

Table 7.5 Boat and Barge Passage Through the Illinois Waterway Locks in 2001

Lock Name	Month	Commercial	Recreational	Other	Total tons
Lockport	May	247	120	24	563,512
	June	246	167	29	1,358,209
	July	248	246	6	1,391,360
	August	247	200	15	1,365,849
	September	248	246	25	1,508,708
	TOTAL	1,236	979	99	6,187,638
	Season average	247	195	20	1,237,528
Brandon Road	May	247	156	39	1,356,368
	June	242	204	23	1,354,788
	July	245	354	14	1,415,960
	August	245	319	17	1,427,404
	September	237	283	20	1,523,856
	TOTAL	1,216	1,316	113	7,078,376
	Season average	243	263	23	1,415,675
Dresden Island	May	247	262	23	1,612,186
	June	232	513	22	1,530,496
	July	248	587	12	1,683,457
	August	250	588	20	1,739,073
	September	247	557	16	1,722,730
	TOTAL	1,224	2,507	93	8,287,942
	Season average	245	501	19	1,657,588
Marseilles	May	243	268	14	1,787,951
	June	214	416	22	1,710,473
	July	229	671	5	1,914,036
	August	246	618	61	2,000,573
	September	225	557	21	1,803,183
	TOTAL	1,157	2,530	122	9,216,216
	Season average	231	506	24	1,843,243
Peoria	May	352	254	20	3,210,839
	June	278	57	28	2,701,174
	July	305	629	32	2,873,116
	August	295	541	36	2,669,577
	September	298	420	29	2,500,279
	TOTAL	1,528	1,901	145	13,954,985
	Season average	306	380	29	2,790,997

Conflict Between Recreation and Navigation

The conflict of recreation with navigation is most severe in the Brandon Pool. The conflict is due to the physical restriction of the navigational channel constituting most of the Brandon Road Pool. The width of the waterway is too narrow for safe simultaneous waterborne recreation such as water skiing and navigation. Kayaking and passage of recreation boats is possible with caution but it is hampered by access. Currently, there are no public landings on the Brandon Pool and access is prevented by railings and vertical banks. The nearest public landings and river access for small boats are on the Des Plaines River upstream of the confluence with the CSSC. As stated before, swimming in the Brandon Pool waterway should not be allowed.

However, the City of Joliet is planning to install a boat launch in the near future. The facility is proposed on a 10 acre city river Bicentennial Park parcel north of Ruby Street. This facility will have three launch ramps, and car/trailer parking for 25 vehicles. A restaurant is also proposed for the site. The boat launch is a part of the City's river front development. The City has applied for state grants and permits for the project to be constructed in 2002 (Personal communication, Don Fisher, City of Joliet Planning Department). The above construction of the boat launch is a part of the City's effort to redevelop the downtown around river front recreation, entertainment and downtown housing. The City plans to complete the entire river walk by 2006. Most of the upland portions are already in place. The master plan is making the river a main focal point for the downtown area. The perception of water quality by the Joliet citizens has apparently improved to a point that the City is sponsoring fishing tournaments and citizens have noticed recent improvements in fish diversity. The City also sponsors several festivals each year along the river. Under the City's master plan they will not encourage swimming (full body contact); however, the city wants to provide opportunities for more fishing and recreational boating activities. A private boat launching facility is planned near Jackson Street. The facility will be a part of a townhouse development with attached boat slips.

Navigation may not be impeding the recreational opportunities in the Dresden Island Pool and limited recreation is feasible in most sections. Therein navigation is restricted to the deep central channel and the navigation channel is marked by buoys.

A question of reversibility should be addressed. It could be argued that the river could be renaturalized, navigation reduced or replaced by other transportation means, etc. The destiny of the neighboring and abandoned Illinois - Michigan canal reminds us that such work and river modifications are not eternal. However, today the Illinois Waterway is one of the premier inland waterways in the nation and abandoning navigation in it is not possible in the short and long run and most likely would result in an adverse wide spread socio-economic impact, interrupting the navigation connection between the Great Lakes (Atlantic Ocean), Mid-America grain region, and Gulf of Mexico. The Clean Water Act specifically states that the water quality standards must recognize navigation as a beneficial use.

Existing Use

AquaNova International has contacted by phone several marinas and bait shops, government institutions and personnel located on or near the Lower Des Plaines River. In addition, numerous web sites were also viewed. Respondents to the inquiries included:

> GPO (game warden), Will county sheriff patrolman on the river, four marinas (downstream of I-55 bridge on Dresden Island Pool), Will County Resources Management representative, DNR Des Plaines Wildlife Refuge Area representative, Park Ranger, Lower Des Plaines River Ecosystem Partnership representative, Site supervisor for the Channahon State Park, owners of local bait shops, several citizens from Lockport, Joliet and Dresden Locks.

Each respondent was asked the following questions:

1. How is the Lower Des Plaines River used for recreation?
2. How many recreational boats are there in a summer week?
3. What type of recreational boats?
4. Have you observed swimming? Other recreational activities or sports on the river?
5. Do you think that the recreational use would increase if the water quality improved? How?
6. Do recreational boats use the locks?
7. Would the use change if there was less commercial barge traffic?

Summary of Responses

Question #1

The river is used for both commercials and recreational boat traffic. Five respondents stated that there is a lot of transient traffic of large boats between Lake Michigan and the Gulf of Mexico ports. Recreational boats stay in the investigated sections of the Lower Des Plaines River.

Question #2

50% of respondents did not know how many recreational boats pass the river, positive answers ranged from 20 - 30 to more than 500.

Question # 3

All respondents noticed recreational boats. Size of the boats were ranging from small fishing boats to large yachts.

Question # 4

No swimming was observed in the Brandon Pool. Only four out of 18 respondents observed occasional swimming in the Dresden Island Pool; however, mostly in the section of the pool downstream of the I-55 bridge. Some marinas (downstream of I-55 bridge) reported that people are reluctant to swim in the river because of their perception of sewage pollution of the river. Swimming has been observed mostly from boats. Other activities such as water skiing and tubing have been observed, for example, by the lock operator of the Dresden Island Lock (downstream of the I-55 bridge) or by the Will County sheriff patrolman and GPO.

Question # 5

All respondents answered affirmatively, i.e., recreational use would increase if water quality improved. However, the perception of "bad" water quality was strong. The recreational uses that would most likely improve are, in the order of positive response, fishing, canoeing, bird watching, and swimming.

Question # 6

The traffic is heavy during summer months. The numbers provided by the US Army Corps of Engineers do not include boat traffic that does not pass through the locks (i.e., they launch the boat and remain in the Dresden Pool).

Question # 7

The respondents were about evenly split. 8 said that the reduction of commercial barge traffic would have no impact on the recreational use, 9 said that it would.

Recreational boat traffic information for the April-September, 2001 period was provided by the US Army Corps of Engineers, Rock Island, IL (see also Table 7.2):

Lock	Number of Recreational Boats Passing through the Lock
Lockport	1,031
Brandon Road	1,284
Dresden Island	2,622

Planned Use of the Brandon Pool

The survey indicated that primary recreation is not an existing use in the Brandon Pool. Swimming in the Dresden Island Pool is infrequent and occurs mostly in the section downstream of the I-55 bridge. This type of use cannot be characterized as existing primary contact recreational use.

However, the proposed river park and downtown development in Joliet will necessarily push for water quality improvements that would provide for non contact recreation even in the Brandon Road Pool. The city and its sanitation department should be responsive to a call for disinfection of their effluents to meet the water quality (in the Dresden Island pool) that would provide for such recreation.

Overall Assessment of Use Attainability for Primary and Secondary Recreation and Proposal for Standards

New Standards based on the USEPA(2002) Draft Guidelines

The USEPA has been promulgating the *E.coli* and enterococci-based standards since their issuance in 1986. So far, less than half of the states have complied. The new USEPA(2002) draft guidelines gave the states more flexibility in the choice of the risk on one side but indicated that the USEPA will be less flexible as to implementation. Also, based on the new Clean Water Act Amendment dealing with beach pollution passed by Congress on October 10, 2001 (BEACH Act), the State of Illinois may have to adopt the new standards by October 10, 2004. Thus, it does not make much sense to try to develop site specific standards for the Lower Des Plaines River using the old fecal coliforms indicator numbers. This study proposes to adopt the new standards based on the preceding analysis of the fecal coliforms data that served as a reliable surrogate. The *E. coli* group is a subgroup of the fecal coliforms group and literature studies indicate a close correlation between the two groups. In essence, if the fecal coliforms measurements meet or are close to the new numeric criteria based on *E. coli* there is a scientific certainty that the corresponding *E.coli* measurement would also meet the standard.

Formulation of the new *E. coli* based standards begins with a state accepting a risk of waterborne illnesses caused by primary contact. The acceptable risk (Table 7.1) varies between 8 illnesses/1000 swimmers to 14 illnesses/1000 swimmers. If existing water quality meets this range it would be prudent to select the risk at this level. **The risk is the primary standard and it is up to the discretion of the state to select the risk within this range.** The magnitude of the *E. coli* standard is then related to the risk. This flexibility is a step forward from the rigid single risk standard (8 illnesses/1000 swimmers) that was a foundation of the previous general use standard using fecal coliforms as indicators (i.e., geometric mean of 200 cfu/100 mL and 400 cfu/100 mL not to be exceeded in more than ten percent of samples during any thirty day period).

This study recommends adopting the new bacterial standards that use *Escherichia coli* as indicator organisms that are based on the level of risk acceptable to the State of Illinois for the reaches of the Lower Des Plaines River.

Brandon Pool (RM 291.0 - 286.0)

The following suite of factors impede water borne recreation and attainment of the primary recreation standards in the Brandon Pool:

1. *Actual use:*
   The primary contact recreation is not an existing use. Swimming in the Brandon Pool has not been observed. Secondary contact water recreation is limited to fishing and aesthetic enjoyment. Larger recreational boats are mostly passing the pool and the people do not engage therein in contact recreational activities. Extensive water skiing and power boating may not be possible because of the barge traffic and narrow channel.

2. *Existing water quality:*
In the Brandon Road and Dresden Island Pool, the river is effluent dominated and its water quality is impacted by the effluents and CSOs from the Metropolitan Chicago area that has a population of more than 9 million. Waste water effluent and CSOs from Joliet cannot reach Brandon Pool because the elevation of Joliet is below the water surface elevation in the Brandon Pool. The existing bacterial water quality does not meet the Illinois standard for primary contact recreation.

3. *Water quality potential:*
Reference water bodies in the Illinois River system that are minimally impacted by urban development do not attain the current general (primary contact) use standards for maximal FC densities (Reason 1; 40 CFR 131.10(g)). Bacteriological quality could be improved by disinfection of effluents from Joliet and other WWTP located on the Hickory Creek (if they are not currently disinfecting) and additional planned control of CSOs by TARP and by best management practices for urban runoff. The potential water quality could meet the EC based standards for swimming derived from a higher, yet, acceptable risk level.

4. *Access:*
Water access to the river along most of Brandon Pool is prevented by steep concrete and sheet pile embankments with railings. Until 2002 no public or private (marina or boat landing) access was located on the Brandon Pool. However, a boat launch is being considered and most likely will be built.

5. *Recreational facilities:*
The Bicentennial Park in Joliet is the main recreational facility on the Brandon Pool. The park has not been designed or developed for primary recreation and has no facilities for such activities. The park is used for cultural activities, picnicking, and watching the river. As pointed in Item 4. above, a boat launch is being built. The City is making the river its focal point for downtown development and promotes noncontact recreational opportunities.

6. *Safety consideration*:
The water body serves as a major shipping lane that occupies the entire width of the pool. The shipping and steep banks of the navigation channel prevent swimming and severely restrict other non-contact recreation. Swimming and water skiing is dangerous and could result in drowning and collisions with barges that occupy a large portion of the channel.

Recommendation

Primary contact recreation is not feasible at the Brandon Road Pool and should not allowed. There are two options open to the IEPA and IPCB for the designated recreation use and microbiological standards.

Option I - No Recreational Use of the Brandon Pool

Rationale: Primary contact recreation is not an existing use and is not possible due to physical features of the pool and interference with navigation. The water body is fenced off and swimming would be dangerous to the swimmers. Swimming also should be discouraged because of the effluent domination of the river. Noncontact recreation is limited to recreation boats passing through the pool and to aesthetic enjoyment of the river by citizens and visitors of Joliet.

The IEPA could recommend to the Illinois Pollution Control Board to prohibit the recreational use of the Brandon Pool with the exception of sightseeing, recreational boat passage and recreational fishing (with a fish consumption advisory). This use prohibition would have to be periodically reassessed in accordance with the Clean Water Act and water quality standards regulations. This recommendation would be based on irreversible physical impediments to the primary and secondary recreation in and on water due to navigation and physical features of the Brandon Pool.

This prohibition of the use would also be based on the argument that passage of boats between the Lockport and Brandon Road Locks through the Brandon Pool (average 7/day) cannot be truly considered "recreation". Passengers generally do not engage in recreational activities (water skiing, power boating, etc.).

Option II - Secondary Use

Rationale: Recreation by boating requires protection by secondary use standards.

Selection of the Risk and Standard

Under the new approach to assigning the recreational use and the corresponding standards, the first step is defining the appropriate risk for the water body. Because the physical irreversible attributes, navigation and effluent domination, primary contact recreation is not proposed and is discouraged. However, recognizing the fact that recreation boat traffic through the Brandon Pool is occurring, and the boat launch will be built, the designated use of the pool would be secondary non contact recreation. The risk for such use should be higher than the risk for primary contact recreation that was recommended between 8 to 14 illnesses/1000 swimmers. This UAA proposes to establish a standard that would recognize the fact that primary contact either is not existent or would be very rare and incidental. This standard would be five times 548 cfu of E. Coli/100 mL which is five times the criterion based the highest primary contact risk of 14 illnesses/1000 swimmers. The standard is then 2740 cfu/100 mL of *Escherichia Coli* indicator organisms measured as geometric mean of samples. No single maximum standard is proposed.

This water quality, expressed by the fecal coliforms densities, is existing, i.e., the currently measured geometric mean of 350 fecal coliform bacteria cfu/ 100 mL is greatly below the proposed secondary use standard of *E. coli.* The AquaNova/Hey Associates team feels that, in the next standard evaluation cycle, the agency could adopt a standard that would be based on a smaller risk.

For example, the water body could meet a secondary standard based on the value five times the lowest risk (8 illnesses/1000 swimmers) that is 630 EC cfu/1000 mL; however, the difference between the proposed standard and current geometric mean provides a margin of safety. Because the *E.coli* densities must be less than that of fecal coliforms (*E. coli* is a part of fecal coliform group) it can be stated with a great scientific certainty that the current water quality would meet the proposed *E.coli* geometric mean standard for secondary recreation and water quality at this level is existing.

Whereas

- the fact that the City of Joliet is planning to use the river as the central focal point of the City's downtown development, including installation of public boat launch, and
- the current water quality, expressed in fecal coliform densities would most likely meet the proposed *E. coli* standard

Option # 2 is recommended for implementation as the site specific standard for the Brandon Pool. Due to the physical restriction of the pool, navigation and effluent domination of the flow, primary contact recreation cannot be recommended and should be discouraged by the City of Joliet and the Illinois EPA even though a high risk *E. coli* standard for primary contact of 14 illnesses/1000 swimmers would most likely be attainable[2]. The secondary use designation still may provide some protection to accidental swimmers.

To put the risk in a perspective, assume that 50 accidental bodily contacts will occur during one year in Brandon pool. By extrapolation of the risks in Figure 7.1, the secondary use standard would correspond to the risk of gastrointestinal illness of 21 case/1000 swimmers. The estimated incidence of gastrointestinal sickness would be 21 x 50/1000 = 1.05 or less, about one sickness in a year. The most common water contact gastrointestinal sickness is diarrhea.

Using enterococci as indicator organisms is not recommended because they are primarily used for marine beaches.

Dresden Island Pool (RM 286.0 - 271.5)

The Dresden Island Pool extends from the Brandon Road Dam and Lock to Dresden Island Dam. With respect to the designated use, the Dresden Island Pool is divided by an artificial boundary at I-55, with the upstream of I-55 designation being the indigenous aquatic life and secondary contact recreation (without a standard for pathogens) and the downstream section, called the "five miles stretch," having general use and primary contact recreation use designation. This legal division makes little sense because neither the public using the pool for recreation nor fish living in the pool may be aware of it and there is obviously no sharp boundary in water quality between the two

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[2]It should be noted that current geometric mean of 350 FC cfu/100 mL is below the primary contact standard based on the of 13 illness/1000 swimmers, which is 429 E. Coli cfu/100 mL. However, the Brandon pool has been found as unsuitable for swimming.

sections. However, with respect to the designated use for recreation and pertinent water quality standards, the following factors have been presented and documented in the preceding sections:

1. *Actual (existing) use:*
   The Dresden Island Pool recreational use is primarily downstream of I-55 where four marinas and public landings are located. The pool is used for fishing, boating, water skiing and also occasional swimming was observed. The sections downstream of RM 283 have natural beauty assets. However, there some are sections of the pool where contact and non contact recreation would be restricted due to navigation.

2. *Existing water quality:*
   AquaNova/Hey Associates evaluation has found that the section between the Brandon Road Dam and I-55 bridge meets most of the water quality standards characterizing the general use. The biological character was found as marginal, below the threshold for the general use, but not much different from the section of the Dresden Island Pool downstream of I-55. These concerns do not prevent designating the entire reach as general use (see Chapter 8).

   The Dresden Island river flow is still effluent dominated by distant MWRDGC discharges and wastewater effluents and sewer overflows from Joliet that are directed into the Dresden Island Pool. The impact of Joliet on the Dresden Island Pool is significant and increases bacterial densities in the pool. Currently (year 2002), the densities of fecal coliforms are 3 to 4 times higher than the standing Illinois General Use Standard based on the geometric mean of 200 FC cfu/100 mL.

3. *Water quality potential:*
   Following the evaluations presented in this document and in this chapter, tremendous progress has been made in improving the water quality and additional improvements can be expected in the future; however, the improvements in frequency of the recreational use may not be significant because of the perception of the users about the water quality (effluent domination of the water body). Bacterial quality can be improved by reinstating disinfection of upstream effluents, especially those from the Joliet East and West plants that would be environmentally sensitive and not harming the aquatic biota or public health. Control of urban runoff in Joliet should also be considered. The required reduction of bacterial densities is about 50 % plus a margin of safety. The impact of distant MWRDGC plants discharging into the Chicago waterways and Des Plaines River upstream of Lockport would be less noticeable.

   The Illinois General Use maximum standard of 10% or less of samples being allowed to exceed 400 fecal coliforms cfu/100 mL is not attainable.

4. *Access:*
   In the section upstream of the I-55 bridge access is somewhat limited by a lack of public landing and marinas and there are no beaches. There are four marinas and a public landing in the more natural and less inhabited section downstream of the I-55 bridge. The lower pool

between RM 283 and the I-55 bridge has a potential for increased recreational use, including contact recreation, mostly from boats and water skiing. Building a boat launch in Joliet may increase recreational use in the Dresden pool because the boaters will gravitate to more desirable recreation in the Dresden Island Pool.

5. *Recreational facilities*
There are four marinas of the Dresden Island Pool, one of them right downstream of the I-55 bridge. Downstream of RM 283 the river is surrounded by forests and natural lands. Most of this land is privately owned. The Empress Casino is operated as a resort that would benefit from expanding the recreational opportunities. The area has recreational potential. There is a potential for developing most of the Dresden Pool as a recreational area for the citizens of northeast Illinois.

   Building a boat launching facility in Joliet may add to the frequency of boating in the Lower Dresden Island pool.

6. *Safety considerations.*
Barge traffic does represent a safety concern in some sections; however, the river is sufficiently wide enough to allow both recreation on water and commercial barge traffic with safety precautions of both users.

Selection of the Risk

The factors that would prevent primary use in Dresden Pool, such as it was in the Brandon pool, are not present. Therefore, primary contact recreation should be protected by the standard. However, the use of this water body for primary recreation will be marginal at best and mostly incidental (e.g., occasional falling from water skies). Incidences of swimming in the pool will be much less than in the other Illinois waters. Beaches for swimming should not be developed in this reach at this time.

Thus the proposed risk corresponds to the highest risk for primary recreation that the state can select without a UAA. It is up to the discretion of the state to select the risk of 14 illnesses/1000 swimmers. In the future the risk can be lowered as the water quality improves. Also, it is also up to the discretion of the state to impose a lower risk for the section of the Dresden Island Pool downstream of the I-55. Logically, the entire Dresden Island Pool should have the same standards and will have for most other parameters (see chapter 7).

Recommendation

With respect to the Dresden Island Pool, the Illinois EPA and Illinois Pollution Control Board have two options:

Option I. Extend the primary recreation use and the uniform standard for pathogens to the entire Dresden Island Pool

The Upper Dresden Island pool has natural assets that promote primary recreation, especially in the section downstream of mile 283. On the other hand this stretch of the river also has a relatively high concentration of industrial activities and most recreation will still occur downstream of the I-55 bridge. Nevertheless, the expected frequency of swimming will still be low and frequency of the primary contact recreation will be much less than in the other Illinois streams; therefore, the state may choose a higher acceptable risk. For example, a risk of 14 illnesses/1000 swimmers could be acceptable. This risk implies that if a moderate frequency of swimming in the Upper Dresden Island Pool is, for example, 100 swimmers over a period of 3 summer months, the probability of gastrointestinal illness would be $14 \times 100/1000 = 1.4$ per year ≈ 1/year.

It is also expected that the frequency of the primary use would be characterized as "Infrequently Used Full Body Contact" or as "Marginal Primary Contact Recreation."

The *E.coli* based standard for this level of risk would then be (Table 7.1):

Geometric mean density of *E.coli* 548 cfu/100 mL

The single value maximum is for beach closings and swimming advisories:
From Table 7.1 this maximum value for the risk of 14 illnesses per 1000 swimmers is 2507 E.Coli cfu/1000 swimmers.

Using enterococci as indicator organisms is not recommended because they are primarily used for marine beaches.

The IEPA and the Illinois Pollution Control Board may choose to adopt a lower risk of contacting waterborne illness; this is up to the state discretion.

The FC based standard should be discontinued. Due to the fact that there is a great similarity between the *E. Coli* and fecal coliforms densities and *E.coli* density cannot exceed that of fecal coliforms, continuation of the fecal coliforms based standard does not make sense. In the next year, the agencies and dischargers should focus on developing data bases for E/Coli indicators.
The proposed standards are attainable (with disinfection of Joliet effluents) and would provide adequate protection for contact recreation in the entire Dresden Island pool.

- Abandon the maximum limit of 10% of samples can exceed 400 FC cfu/100 mL that is not attainable in the Lower Des Plaines River and its reference sites and is overprotective based on recent USEPA (2002) draft standard guidelines.

Option II. Secondary Use with Primary Use Protection (Restricted Primary Contact)

Rationale: The river in the Dresden Island Pool is still an effluent dominated water body. Declaring this section of the river as supporting secondary non contact recreation only would give the public a warning to exercise caution and legal protection to agencies. The use of the Dresden Island Pool is also restricted by commercial barge traffic but, in most of its length, not by physical channel constriction and access. The recreationists should be notified about these aspects at boat landings and parks. However, the primary use standards as specified above should be implemented because they are attainable (with modifications specified herein) and infrequent primary contact use such as water skiing and swimming occurs.

The choice between Options I and II is a policy decision that will have an identical impact on water pollution control efforts and clean up of the Dresden Island Pool. Because Option II retains all features of primary use protection, it could be characterized as a subclass of the general use, e.g., "Restricted primary contact" and not "Secondary noncontact recreation."
AquaNova/Hey Associates recommend adoption of Option I. Classifying the use as a secondary contact while primary contact standards would be attainable is not recommended. Secondary use must be evaluated by a UAA every three years while a primary use fully complies with the Section 101(a) of the Clean Water Act and would not require triennial reissuances of UAAs. Secondary use designation would also keep the water body on the TMDL 303(d) list while adopting the proposed higher risks primary contact recreation would remove the bacterial contamination of the Dresden Island Pool of the Des Plaines River from the 303(d) listing[3].

Using enterococci as indicator organisms is mostly for marine waters and is not recommended for the Upper Illinois Waterway.

Because of climatic conditions of the area, the state may consider designation of the recreational uses as seasonal.

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[3] Although the current bacterial densities expressed with fecal coliform indicators may be higher than the proposed *E. Coli* standard, the standard can be met by application of the CWA Section 306 effluent control technologies in Joliet and Hickory Creek and application of economical best management practices for urban runoff.

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CHAPTER 8

MODIFIED IMPOUNDED WATER USE DESIGNATION FOR THE BRANDON ROAD POOL AND USE UPGRADE FOR THE UPPER DRESDEN ISLAND POOL

Introduction

Many water bodies have been modified to serve various purposes other than propagation of aquatic life. The multipurpose use of water bodies is common in the civilized world and rivers have been altered for various uses since the time of Egyptian pharaohs for

- Flood conveyance and control
- Providing habitat for aquatic biota
- Providing for contact and non contact recreation and aesthetic enjoyment
- Providing water for public and industrial water supply and irrigation
- Providing flow for various in-stream uses such as hydro power production
- Navigation
- Providing cooling water for thermal power generation
- Disposal of residual waste loads

The main objective of the Clean Water Act in Section 101(a) is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. Section 101(a) declares aquatic life protection and propagation, and contact and non contact recreation the superior uses to be attained.

In most cases, economic uses listed above were achieved by the physical alteration of the water body such as

- impounding and channelizing the river to provide navigation depth and head for other water works (e.g., for example, hydropower generation, navigation and irrigation);
- periodic dredging of sediments in the natural and impounded reaches to maintain navigation;
- diking and building embankments to control floods and prevent extensive flood damage, especially in congested urban areas;
- man made channels that relocated the former bodies or were built as completely artificial water bodies (e.g., California Water Project canal, Chicago Sanitary and Ship Canal, and the Brandon Pool of the Lower Des Plaines River).

Figures 8.1 through 8.3 show examples of modified water bodies that may require a special use designation based on UAA Reasons #4 and #5 specified in Box 1.1. A new use, or modified sub use designation, must be based on the optimum ecological potential of the water body that would still meet the goals of the Clean Water Act.

Throughout the years, these water bodies have become a part of the landscape and are being used or could be used for activities such as fishing, limited recreation and other uses (Figure 8.4).

Commonly, they are connected to natural streams, they are near or a part of population centers, and they are subjected to government jurisdiction and responsibility derived from the Commerce Clause of the Constitution. Therefore, they require protection and compliance with Section 101(a) goals of the Clean Water Act.

A key feature of the water body that may qualify it for special use designation is *irreversibility* of the physical impediment, or deficiency that prevents the attainment of the designated use. The test of irreversibility should be evaluated based on the UAA criteria that specify in the long run

- the condition cannot be remedied or would cause more environmental damage to correct than to leave in place, and/or
- removing the condition would result in substantial and wide spread adverse social and economic impact.

Navigation and water supply are beneficial uses specifically mentioned in the CWA Section 303 (c) (2) which specifies that for water bodies

> *.....standards should be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, and agricultural, and other purposes, and also taking into consideration their use and value for navigation.*

Therefore, active navigation is a protected use and cannot be removed solely for improving the water quality. However, if navigation is impeding integrity of the water body it should be modified so that the water body integrity is maintained. In the case of the Illinois Waterway, removal of navigation could also cause a wide spread adverse socio - economic impact as shown in Table 7.2 that reported monthly tonnage of cargo passing through the Upper Illinois Waterway ranging from 1.4 to 1.8 million tons (Reason # 6 of the UAA regulation). However, navigation use may not be permanently irreversible as exhibited by the commercial demise of the Illinois-Michigan Canal[1] in the past and present efforts to renaturalize the Missouri River.

Most navigable water bodies could provide conditions for a balanced aquatic life and should be classified with a use commensurate with the Section 101(a) of the Clean water Act; i.e., the General use in Illinois. The purpose of the use designation is not to downgrade the use, but rather to reflect the reality that the biological composition of such water bodies may not be comparable to pristine unimpacted reference streams that form the foundation of the biotic integrity indices. The integrity of these streams should be compared to least impacted water bodies that have the same morphological character, i.e., being impounded and navigable.

On the other hand, a simple fact that a dam was built on the river and interferes with water quality does not make the situation irreversible. Impoundments built decades to more than a hundred years ago for providing head to numerous and later abandoned mills and small power plants became a

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[1]Today, 61 miles of the Illinois-Michigan canal are managed as a park a nature trail.

water quality problem by collecting sediments that, in many cases were contaminated. Such water bodies are prime candidates for restoration (Figures 8.5 and 8.6).

Water bodies that are heavily used for economic purposes may have one common facet. They may need help in order to achieve the optimum water quality goals. Humans have been using these water bodies for many years for economic benefits that may infringe on ecological health. Without management these water bodies would not achieve their ecological potential. Humans should provide management means that would compensate for the effects of physical modification and uses and lead to optimum water uses in agreement with the overall goals of the Act. Such measures may include in-stream or side stream aeration, fish stocking, periodic sediment dredging, nutrient inactivation, etc. A plan for water body management should be following the UAA.

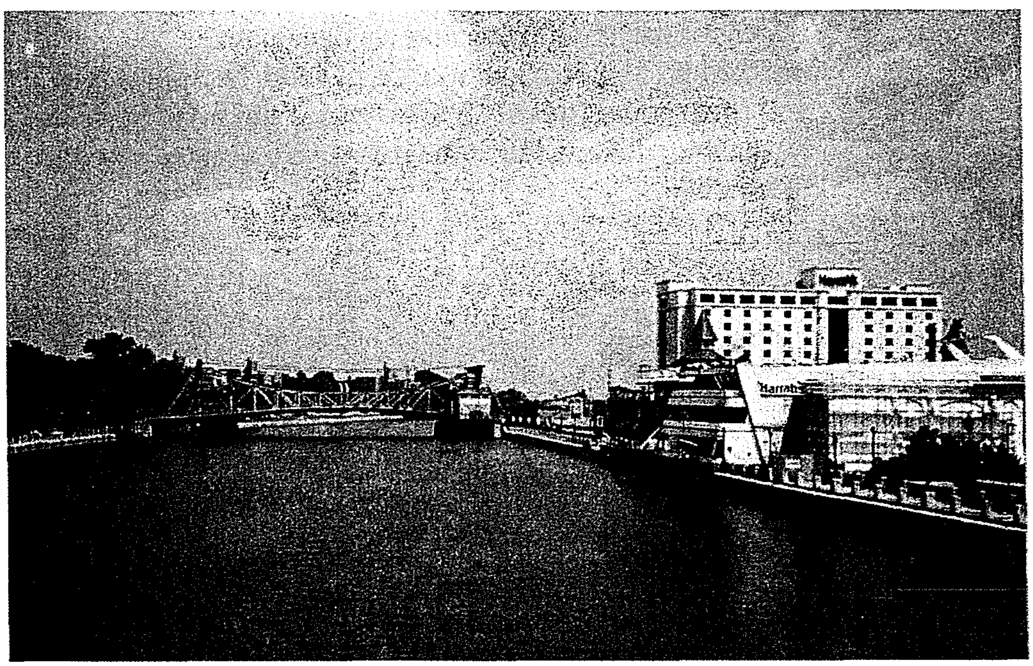

Figure 8.1 **The Lower Des Plaines River in Joliet was converted almost one hundred years ago into a navigation canal with concrete or sheet pile embankments. It is characterized by heavy navigation density.**

Figure 8.2 Milwaukee River in Downtown Milwaukee (WI). The river is constricted by downtown development, is maintained as a navigable channel and has poor habitat and reaeration. Relatively good water quality is provided by pumping lake water into it at a point upstream of the downtown.

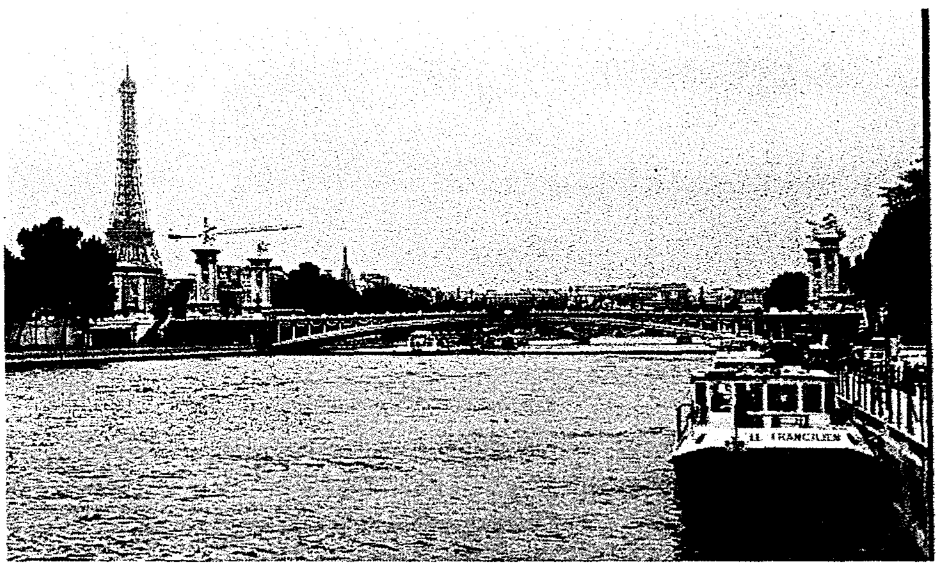

Figure 8.3 Seine River in Paris. One of the grande rivers of the world. Over the centuries it has been constricted by city development and surrounded by historic landmarks. It is characterized by heavy navigation density, both recreational (tourist) and commercial.

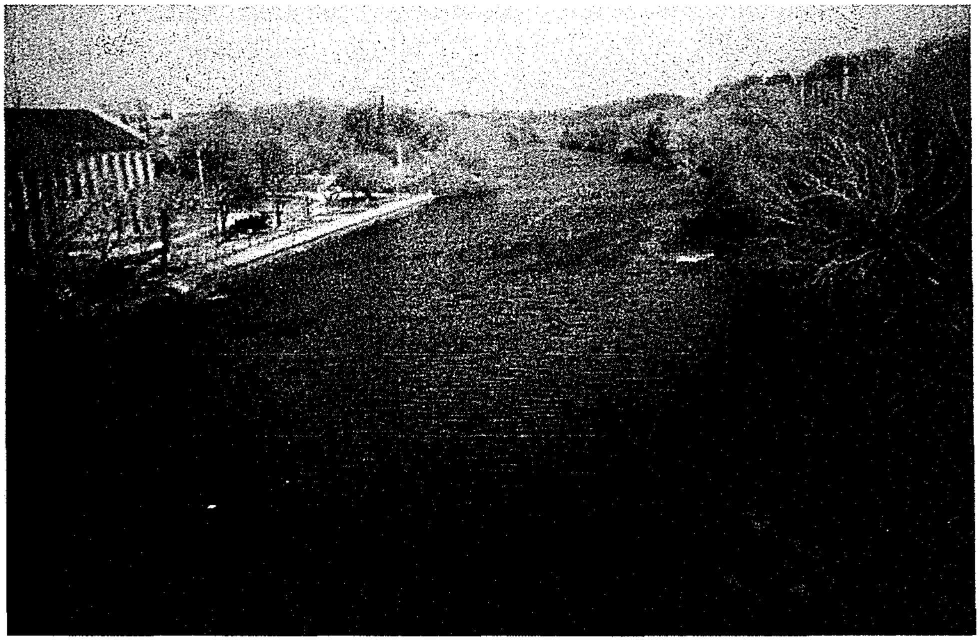

Figure 8.4 North Avenue Dam Impoundment in Milwaukee before 1990. The dam was built more than one hundred years ago to provide head for a navigation canal that was never built. For more than one hundred years it accumulated sediments and became a water quality nuisance, resulting in poor habitat conditions and water quality.

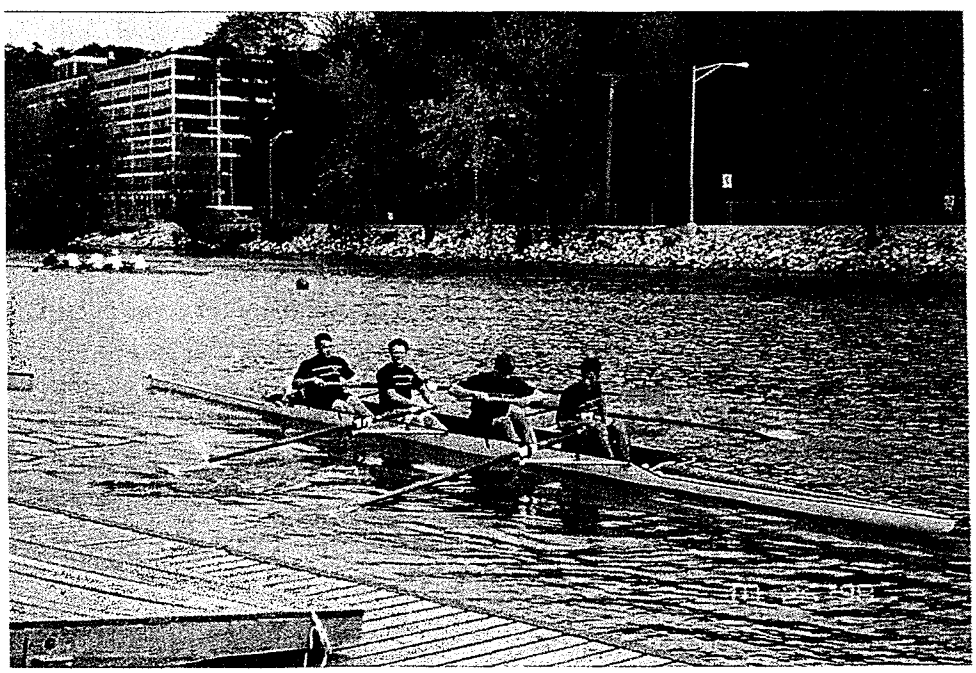

Figure 8.5 Iowa River in Iowa City is a modified urban river that provides good to excellent opportunities for noncontact recreation, fishing and aesthetic enjoyment.

(see Box 1.1). In addition to the chemical parameters evaluation, the UAA must also assess the following:

1. Biotic integrity evaluation detects an unbalanced biotic population (IBI measures indicating less than good-fair ranking);

2. Physical (habitat) integrity quantifies the degree of physical human modifications and impact on the water body that would not provide support for a balanced aquatic biota.

It is recognized that both human physical modifications and impacts (generically classified as pollution but not pollutant) and *pollutants,* i.e., allochthonous discharges from wastewater effluents and other point sources as well as urban and agricultural runoff and other nonpoint sources, can

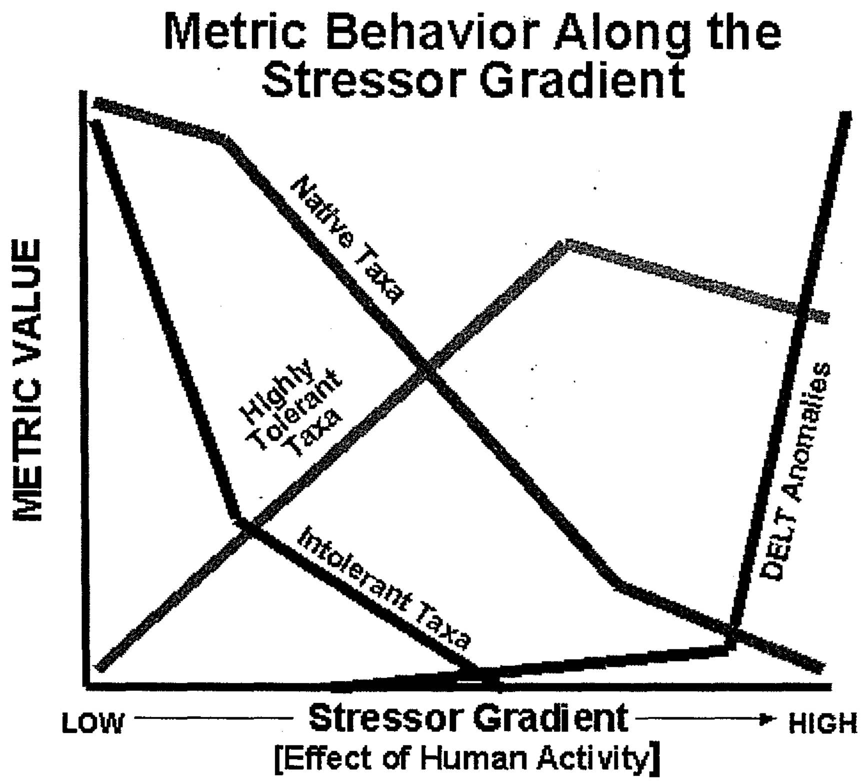

Figure 8.7 Effect of human stressor on the composition of the biotic community (Yoder, 2002)

adversely impact the biotic integrity of the water body. Thus, the biotic and habitat assessment can reveal the waterbody problems caused by pollution while chemical assessment is limited, in most cases, to detection of the impact of pollutants.

<u>Ohio Modified Warmwater Body Designation</u>

This proposed schematic of a special modified impounded use for the Brandon Pool resembles the Ohio modified warmwater body designation shown on Figure 8.8, thus the Ohio designation serves as an example. The State of Ohio system classifies the water bodies using numeric fish and macroinvertebrate IBIs and the physical impairment is described in the narrative terms. For comparative purposes the Ohio water body classification is shown on Figure 8.8 and the numeric limits using Ohio indices are given in Table 8.1. The third category of the Ohio system, water affected by mining, is not included. Note that Ohio does not use the macroinvertebrate index for classification of impounded water bodies that were deemed as unreliable indicators. Following analyses of IBIs on hundreds of Ohio streams, Ohio scientists and regulators realized that impounding a river, even in absence of other pollution, is a stressor that reduces the magnitude of the IBI. Thus, they implemented the *modified warmwater use*. The modified warmwater use has been defined by the State of Ohio as (State of Ohio, Rule 3745-1-07):

Table 8.1 Ohio Biocriteria and Designated Uses

Modified warmwater habitat		Warmwater habitat	Exceptional warmwater habitat
Channel modification	Impounded		
Index of Biotic Integrity (IBI-fish) (Values Different for Five Ohio Ecoregions)			
20 - 24	22 - 30	32 - 44	> 48
Invertebrate Community Index (Macroinvertebrates)			
22	-	30-36	46

"Modified warmwater" - these are waters that have been the subject of a use attainability analysis and have been found to be incapable of supporting and mainataining a balanced, integrated, adaptive community of warmwater organisms due to irretrievable modifications of the physical habitat. Such modifications are of lasting duration (i.e., twenty years or longer).... The modified warmwater habitat designation can be applied only to those waters that do not attain the warmwater habitat biological criteria (Table 8.1) because of the irretrievable modification of the physical habitat.

There are several important facets of the Ohio rule:

1. Nonattainment of the biological criteria due to a physical irretrievable impairment is the key. This implies that if a waterbody with physical features that could classify it for this modified use meets the biological criteria for a higher use then the water body cannot receive the lower use designation. For example, if the Dresden Island or any other pool on the Illinois Waterway meets or could meet the higher water use category or has a potential of meeting it the use cannot be downgraded to a lower modified use even when the physical features of the pools would allow a lower use designation. **Thus, there is no blank modified use designation for all impounded waters.**

2. Designating a water body into this category requires a site specific UAA.

3. Demonstrating attainment of the applicable biological criteria in a water body will take precedence over the application of selected chemical-specific aquatic life or whole-effluent criteria associated with these uses.

4. The macroinvertebrate index is not used for impounded waters.

5. Other pollution such as contaminated sediments, correctable physical impairment (e.g., lack of riparian vegetation), or discharges of pollutants or thermal loads cannot justify the modified use designation. If such impairment occurs, the water body should be put on the (action) 303(d) list for development of the TMDL. Only if the implementations of allocations and actions, identified by the TMDL, cause a wide spread socio-economic impact, can the water use be reclassified.

Ohio's modified warmwater body use also includes a modified primary contact recreational use that would be similar (not identical) to the existing restricted secondary use in Illinois (see Chapter 6).

Habitat Evaluation

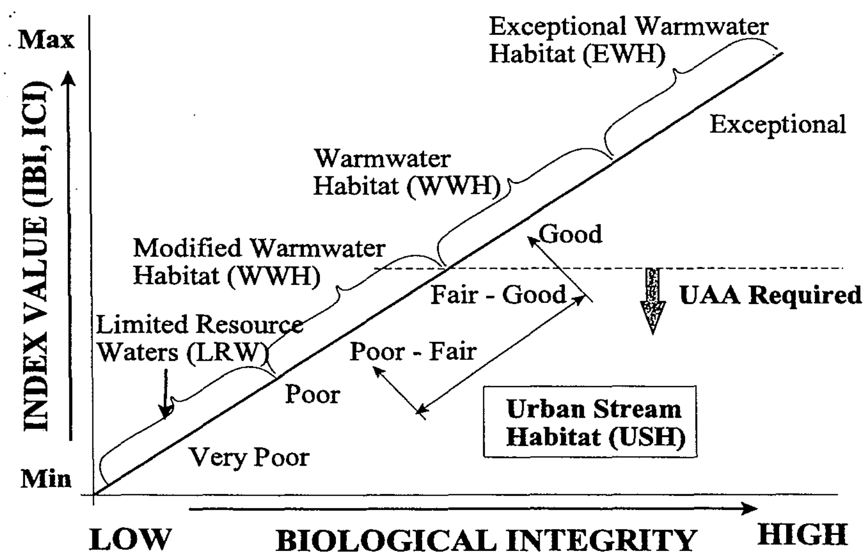

Figure 8.8 Biological community description and quality gradient of Ohio aquatic life uses (Yoder and Rankin., 1999)

Physical features of the Lower Des Plaines River were described in Chapter 4. The typical habitat of the modified impounded warm water body is a constricted channel that has very limited or no littoral zone for early life spawning and propagation. A cross-section of the Brandon Pool that fits this description is shown on Figure 8.9.

This type of cross-section extends almost the entire length of the Brandon Pool. Using the traditional habitat evaluation index (e.g., Rapid Bioassessment Protocols, Plafkin et al, 1989), false reading of "good" habitat may be obtained. For example, such a channel has very "stable" banks due

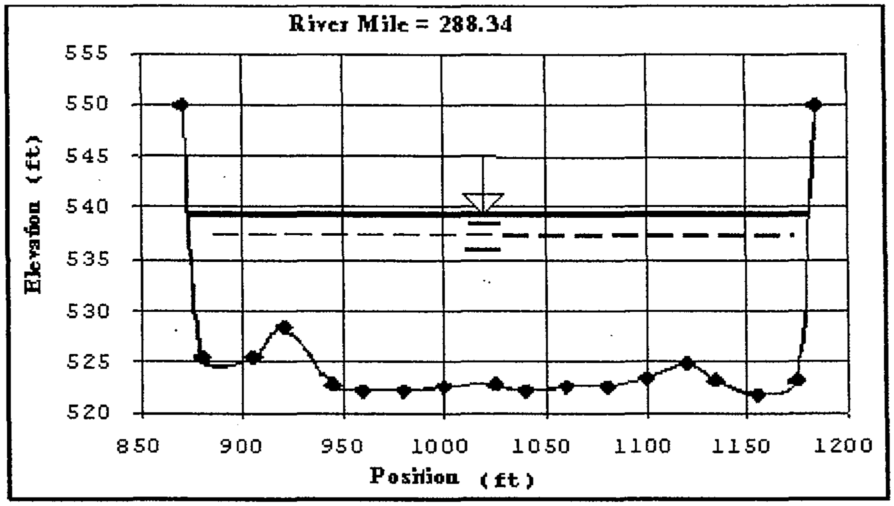

Figure 8.9 **Typical Brandon Pool cross-section in Joliet, IL. The vertical banks consist of sheet pile or concrete embankment. This cross-section does not provide habitat for development of high quality early life species and continuous scouring of the bottom by barges (see also Figure 6.8) prevents development of a quality macroinvertebrate community.**

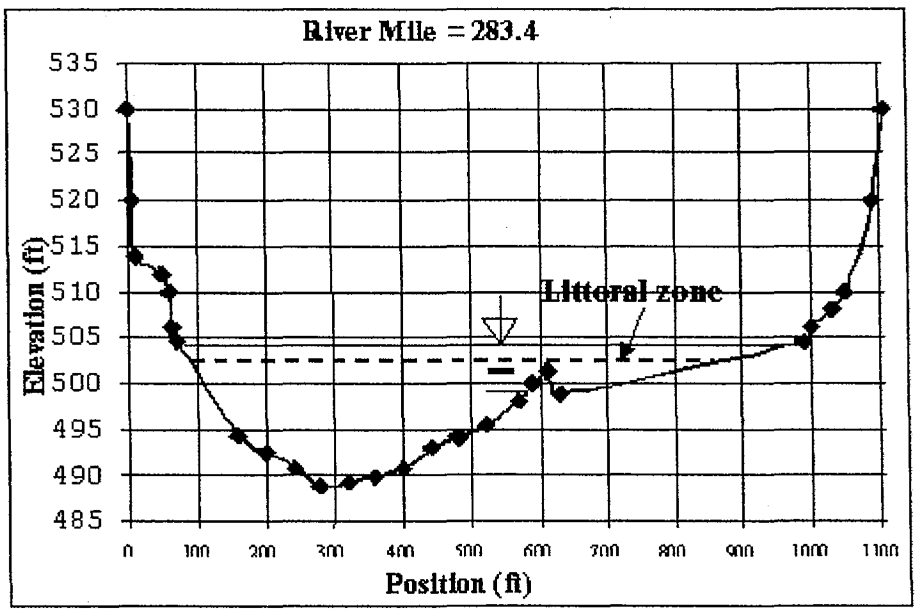

Figure 8.10 **Cross-section of the Dresden Pool showing at this particular section good habitat conditions**

to installation of the artificial embankments and the embeddedness is also "good" because of continuous scouring of the fine sediments by frequent barge traffic.

In contrast, Dresden Island pool has, at least in some parts, reaches that have a shallow littoral zone that provides conditions for good habitat (Figure 8.10). However, it was realized in Ohio that and documented by the USEPA study of the impoundments of the Fox River (see Chapter 6 and a detailed report by Santucci and Gephard, 2003) that impoundment conditions alone can reduce the fish indices of biotic integrity in comparison to the free flowing reference streams. This may imply that the "good" or better ranking by IBI indices developed from, and used for, free flowing streams may not be attainable by impounded streams. Consequently, Ohio classified most of its impounded streams under the modified category. However, a blank categorization of all impounded streams into the modified impounded warmwater body category may not be warranted.

Ecological Categorization and Potential

The first step is to document that early life forms are indeed impeded. This is documented on Figure 8.11 showing total fish and early life forms in the Brandon Road, Upper Dresden, and Lower Dresden Island Pools. In this chart the Upper Dresden Pool is the section of the Dresden Island Poll between the I-55 (RM 277.8) and the Brandon Road Dam (RM 286). The Lower Dresden is the "five mile stretch" between I-55 and the confluence with the Kankakee River. This chart clearly shows that the numbers of the early life forms in the Brandon Dam Pool are very small compared to the Dresden Island Pool, is in spite of the fact that the Des Plaines River upstream from the Brandon Pool has an excellent habitat (Figure 8.12) and much higher numbers of species. The early life species found in the Brandon Pool are incidental and pass through the pool but cannot propagate

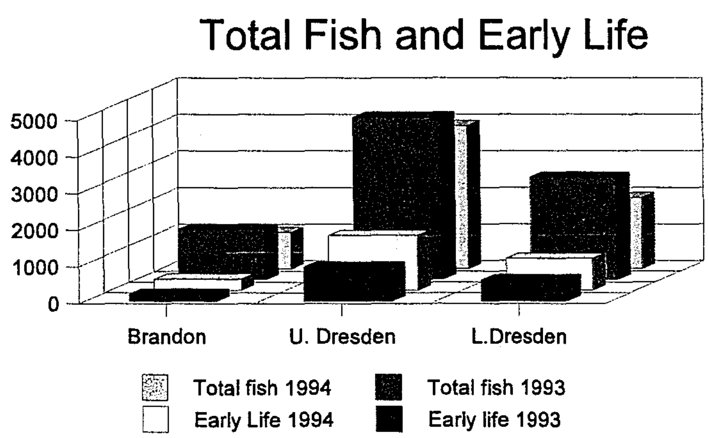

Figure 8.11 Total fish and early life counts in the three pools of the Lower des Plaines River (Data Commonwealth Edison Study)

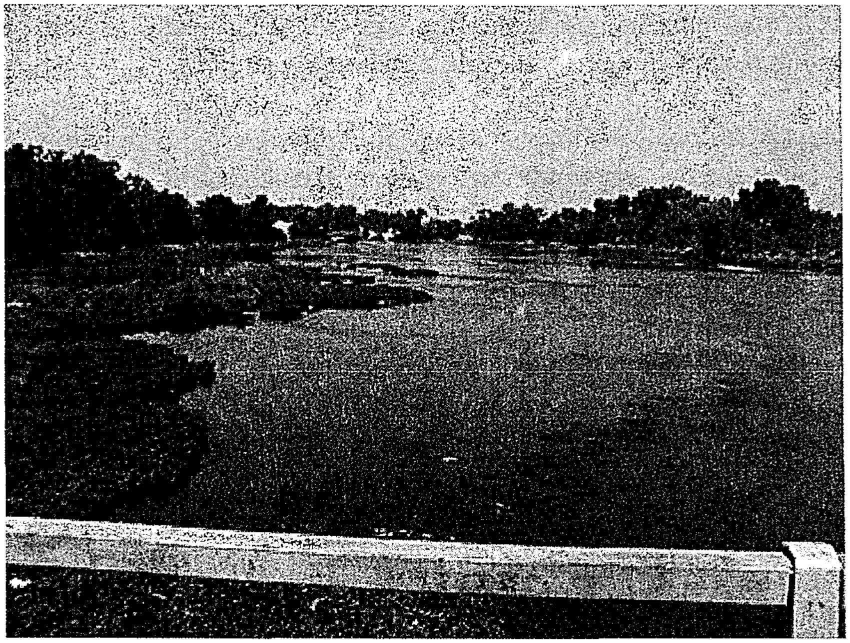

Figure 8.12 Des Plaines River upstream of the confluence with the Chicago Sanitary and Ship Canal in Lockport. The river has very good habitat conditions such as pool and riffle sequence.

because of the physical characteristics of the pool. However, the Brandon Pool also suffers from lower dissolved oxygen levels thus the effects of DO and habitat on early life forms may be symbiotic. Acknowledging the fact that the physical features of the Brandon Road Pool (see Chapter 3 and Figure 8.1) prevent development of early life, a DO standard commensurate with early life form absent can be implemented as specified by the federal DO criteria (USEPA, 1986). In a recent precedent setting ruling, the Illinois Pollution Control Board has adopted the federal criteria for ammonium that also consider the Brandon Road Pool as a water body where early life forms are absent (see the next section on DO and other standards).

The effect of impoundments on the ecological integrity was confirmed by the research on the Fox River (Santucci and Gephard, 2003). In this research fish IBIs were evaluated 0.5 km above (UP) and below (LO) the dams. The upstream measurement reflected impounded conditions, downstream was a naturally flowing channel. The IBIs for the Lower Des Plaines River and comparison with several reference impounded Illinois streams, including the Fox River experiments, were shown on Figure 6.7. The difference between the upstream and downstream sections on the Fox River on Figure 6.7 were consistent and amounted to average IBI reduction due to impoundment of about 12 IBI points. It could be seen that the lower Dresden Island Pool below the I-55 has IBIs that are statistically undistinguishable from the impounded Fox River. There are obvious differences between the Brandon Road Pool and the impoundments on the Fox River. If the stresses in the Upper Dresden Pool (RM 277.8 to 286) delineated in Chapter 2 (primarily temperature) are reduced, attainment of the Fox River ecological goal is realistic. Since the Fox River has been classified as "general use," the same use designation would be appropriate for the Dresden Island Pool and the ecological potential of the Dresden Island Pool could be similar to other impounded larger rivers of Illinois.

However, the Dresden Island impoundment of the Lower Des Plaines River cannot meet the IEPA integrity criterion that is applicable to wadeable free flowing streams.

Following the analysis included in Chapter 6, reasoning behind the Ohio's modified water body classification and using the best impounded and channelized water bodies and not wadeable small headwater streams as references, this specific form of general use can be extended to water bodies that have smaller IBI values. From Figure 6.7, it appears that an IBI of 30 would be a reasonable reference goal for "good" navigable riverine impoundments in Illinois, instead of 50 derived from IBIs of the reference wadeable stream. Therefore, using the same proportions as in the original ranking of IBIs (Karr et al., 1986; Rankin et al., 1990), an impounded water body with consistent IBIs at or above 30, or having a potential of meeting this value may be classified as a general use (impounded) water body. This would lead to a classification of the water bodies as shown on Figure 8.13.

The IBIs for the Brandon Dam Pool are lower and outside of the range that could be classified as potentially "general" use. Brandon Road Pool does not provide conditions for early life forms development and occurrence of these forms is incidental, originating from the upstream Des Plaines River and passing through the pool.

Under the proposed classification shown on Figure 8.13, impounded water bodies that have good to fair habitat conditions such as shallow litoral and backwater refuge areas could be classified as "general use (impounded)." This category is appropriate for the Dresden Island pool. Chapter 6 found that after the habitat quality of the Lower and Upper Dresden pools are similar and because and the lower pool has a General Use classification, considerations should be given to extending the (modified) genberal use to the entire Dresden Island pool. Only waterbodies that are found through a UAA to have physical features and navigational activities that prevent early life spawning, propagation and development would be classified as "modified impounded use." The major reason for this separation is the separation of early life present or absent categories in the US EPA (1986) standing criteria (and Illinois WQS for ammonium) that allow relaxing of the DO, ammonia and some other standard in early life absent situations. The Brandon Road Pool has the characteristic of the modified impounded water body with early life absent and could receive the site specific modified impounded use designation.

From this discussion it follows that, using the best impounded and channelized water bodies as a reference, for example the Rock and Green Rivers, and not wadeable small headwater streams (e.g., the Mackinaw River), this specific form of general use can be extended to water bodies that have smaller IBI values. From Figure 6.7 it appears that IBI of 40 would be a reasonable reference goal for "excellent to good" riverine impoundments in Illinois, instead of 60 derived from IBIs for reference wadeable stream. Therefore, using the same proportions as in the original ranking of IBIs (Karr et al., 1986; Rankin et al., 1990) an impounded water body with consistent IBIs above 30 or a potential of meeting this value may be classified as a general use water body. It should be noted that the original ranking of streams recognizes that the optimum IBI for impounded (channelized) streams is 40 to 44. This would lead to a simplified two-dimensional evaluation of the water bodies such as shown on Figure 8.13.

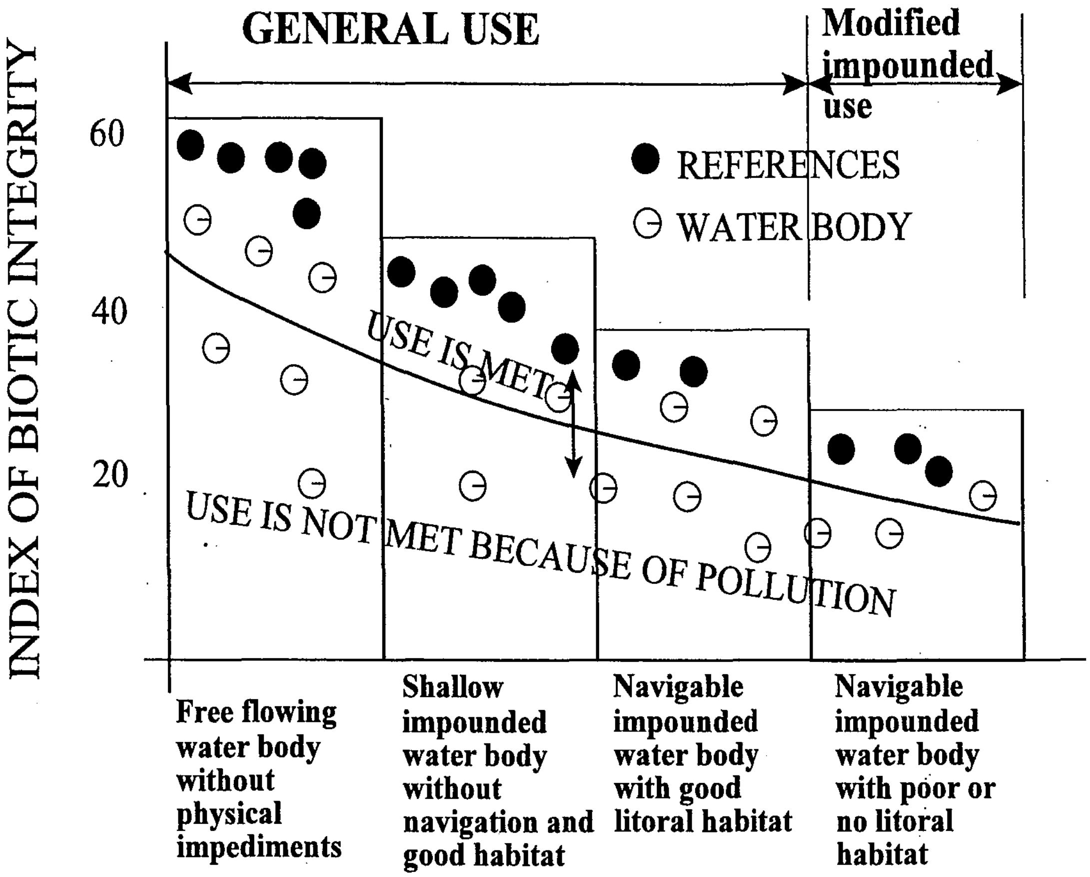

Figure 8.13 Impounded river classification

Under the classification proposed on Figure 8.13, impounded water bodies that have good to fair habitat conditions such as shallow litoral and backwater refuge areas would be classified as ìgeneral (impounded) use.î Only water bodies that are found through a UAA to have physical features and navigational activities that prevent early life spawning, propagation and development would be classified as ìmodified impounded use.î The major reason for this separation is the separation of early life present or absent categories in the US EPA (1986) standing criteria that allow relaxing the DO (USEPA, 1986) and ammonium (USEPA, 1999)[3] criteria and some other standards in early life absent situations. The Brandon Road Pool has the characteristic of the modified impounded water body with early life absent and could receive the site specific modified impounded use designation.

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[3]The Illinois Pollution Control Board had adopted the proposed amendments to ammonia nitrogen standards, which are consistent with the USEPA (1999) criteria. In the Brandon Road Pool, the "early life stages absent" will be used for the entire year to calculate the ammonium standard.

Development of Standards

The impounded or channelized streams that are currently classified in the indigenous aquatic life and secondary contact category would be upgraded to the general use category and assigned water quality standards commensurate with the Illinois General Use unless a UAA, by invoking one or more UAA reasons, justifies a downgrade of the use and/or the standards. For example, Reason #1 of the UAA regulation (40 CFR 131.10(g)) specifies that if the natural/background ecoregional and/or reference water quality are below the established standard the standard could be based on the ecoregional water quality (e.g., 10 percent above [priority pollutants] or below [dissolved oxygen] the natural/background value). Reason #4 deals with the irreversible (in a log term) man-made physical impairment and Reason #5 allows to modify the use if, for example, the water body is lacking substrate or other conditions needed for development of a balanced water biota.

The modified impounded warmwater use classification for the Brandon Dam Pool of the Lower Des Plaines River affects the magnitude of some chemical specific water quality standards that will be different from the general use standards. The standards for the proposed modified site specific Brandon Pool use are based on the consideration of irreversible physical impairment of the water body and are formulated in the ecoregional context. Consequently, impairment of the ecological integrity solely by excessive discharges of pollutants are not considered.

Once a water body is classified by the water body assessment as being impaired and the cause of the impairment is consistent with Reasons #4 and #5 of the UAA regulation, the uniform variance of the standards from the general use is derived from the US EPA (1986) water quality standards for dissolved oxygen and US EPA (1999) for ammonium. Other standard variations are also site specific. However, if a general use standard is met by the existing water quality (e.g., the water body consistently meets the DO minimum of 5 mg/L) the standard cannot be relaxed. A relaxation of the standard, which is attained by the existing water quality, would be against the principle of antidegradation embedded in the water quality standard regulations.

Why the Current Secondary Contact and Indigenous Aquatic Life Standards Cannot be Retained

Arguments and proposals have been made to retain the current Secondary Contact and Indigenous Aquatic Life standards. The exact definition of Secondary Contact is as follows: (Il.Adm.Code Title 35, Subtitle C, Chapter I, Section 302.402)

> *Secondary contact and indigenous aquatic life standards are intended for those waters not suited for general use activities but which will be appropriate for all secondary contact uses and which will be capable of supporting an indigenous aquatic life limited only by the physical configuration of the body of water, characteristics and origin of the water and the presence of contaminants in amounts that do not exceed the water quality standards listed in Subpart D.*

This definition is similar to the objectives of the modified impounded use proposed by this UAA for the Brandon Pool and there are even similarities in wording with the general (impounded) use objectives proposed for the Dresden Pool. However, there are serious inconsistencies between the wording of the objectives and magnitude of some standards for the secondary contact and indigenous aquatic life use (see Table 2.1) that would allow toxic and even lethal conditions to persist in the Des Plaines River. The magnitude of the standards was presented in Chapter 2. The standards that are inadequate for maintaining the indigenous aquatic life are:

Temperature. The secondary contact and indigenous aquatic life standards allow maximum temperatures to reach 100 °F (37.8 °C) and that can legally stay there for an extended period of time (up to 18 days). Literature data and the USEPA (1986) criteria document that the maximum lethal temperature is about 35 °C (95 °F) for the indigenous species exposed to it for a relatively shorter time (1 to 7 days). The chronic standard allows the temperatures to exceed temperature of 93°F for more than 18 days while on site research by the Commonwealth Edison own experts found that in seven days exposure to water temperature of 33°C (91.4 °F) or greater, significant amphipod mortality occurred and a temperature of 34°C (93.2 °F) lasting for seven days was lethal to both amphipod and fish (see Chapter 2 - Temperature). The Commonwealth Edison experts concluded that "*it would appear that the 33°C to 34°C (91.4° to 93.2°F) temperature is the critical range if exposures extend for a period of at least 7 days.*" The margin of safety required by the US EPA (1986) criteria is 2°C (3.6°F) (see Chapter 2 for details). The margin of safety should be subtracted from the critical temperature to arrive at an acceptable standard.

Metals. The standards for some metals of the Secondary Contact use are also in lethal (acutely toxic) zone. The comparison of the General Use and the Secondary Contact use is given below.

Metal	Standards for the Des Plaines River, µg/L		
	General Use standard*		Sec. Contact and
	Acute	Chronic	and Ind. Aquatic Life
Cadmium	2.5(5)	2.3	150
Copper	40(80)	25	1000
Nickel	177 (344)	10	1000
Zinc	260 (520)	46**	1000

* Calculated from on-site hardness
** Federal chronic zinc criterion is about five times larger, see Chapter 2

The numbers in parentheses represent approximate LC(50) (a concentration at which 50 percent of the 5th percentile sensitivity organisms would die). This concentrations was estimated as two times the standard, based on the USEPA standard development guidelines. For cadmium, copper and nickel, the difference between the lethal level and the current standards is more than one order of magnitude. The secondary contact and indigenous aquatic life standards do not provide adequate protection.

The second reason why the Secondary Contact standard cannot be retained is the fact, proven in this UAA, that the values for a great majority of chemical constituents measured during the 1995 - 2001 period (2000 - 2001 for MWRDGC stations) in the investigated reaches of the Lower Des Plaines

River are less than the current general use standards, i.e., most chemical General use standards are already attained.

The current Illinois secondary use has no standards for pathogens that would protect the secondary recreation. Such standards are now required by the USEPA (2000, 2002), even for the secondary use.

Water Quality Standard for Dissolved Oxygen of the Modified Impounded Use

Dissolved oxygen adversely impacts the integrity of a receiving water body in several ways:

1. Low dissolved oxygen concentrations in water are toxic to fish, both chronic and lethal.
   a. Longer low duration DO concentrations inhibit growth and reproduction (chronic toxicity).
   b. Very low DO levels cause fish kills (acute toxicity).

2. Low dissolved oxygen in the water column may change the upper sediment layer from aerobic to anaerobic (typically, a lower sediment layer is devoid of oxygen). This changes the solubility of some compounds and allows a release to the water column. Examples include ammonium/ammonia, phosphates, metals, and hydrogen sulphide. An anoxic or anaerobic upper sediment layer will cause a loss of aerobic benthic vertebrates that are an important component of the food chain. Low DO concentrations in the bottom substrate are also detrimental to spawning.

3. A complete loss of DO in water and/or sediments changes the water body and sediment color to black, which is caused by sulphate reducing bacteria, resulting in the emission of methane and odorous hydrogen sulphide.

The DO levels are affected by the discharges of biodegradable organic matter from point and nonpoint sources, atmospheric reaeration, sediment oxygen demand, nitrification of ammonium and organic nitrogen, and by algal photosynthesis and respiration (Thomann and Mueller, 1987).

Current Illinois DO Standards and Federal Criteria

Table 8.2 provides a summary of the pertinent water quality standards for dissolved oxygen. The standards were derived from the Illinois Water Quality Regulations (Illinois Pollution Control Board, Title 35, Subtitle C) and the federal USEPA (1986) criteria.

It appears that the Illinois General Use DO standard is based on the earlier version of the criteria document published in 1976, the so called "red book"(US EPA, 1976). The criterion of "*a minimum concentration of dissolved oxygen to maintain good fish population of 5 mg/L*" is based, among others, on a 65 year old work by Ellis (1937). This standard was adopted by, and remains in force, in several other states (USEPA, 1988). The 5 mg/L standard was specified in the Illinois standards as an absolute minimum with the exception at flows that are smaller than 7Q10.

The USEPA (1986) revised criteria document (published in yellow covers) relaxed the previous (1976) federal DO criteria. The states were provided with more options and possibilities for site specific standards. Consequently, the current Illinois General Use standard is more rigid in some aspects than the 1986 (and current) federal criterion for early life protection in warmwater receiving waters because the USEPA (1986) criteria document edition for freshwater bodies, added the "early life form absent" category. The criteria in this category are similar to the "indigenous aquatic life" standard for the Lower Des Plaines River and Chicago Sanitary and Ship Canal.

The key decision variables in the formulation of the DO standard in the federal EPA 1986 document are the division of the water bodies into cold and warm waters and categorizing them based on the potential of early life forms present or absent. *The Illinois General Use criteria are similar in magnitude to the USEPA warm water fish species category of the DO limit.* This category is logical for the Des Plaines and other Northeast Illinois water bodies because salmonid cold water fish species are not indigenous to these rivers and could not sustain viable reproducing population.

Table 8.2 Summary of Current Illinois and Federal EPA Dissolved Oxygen Standards

Standard or criterion	Illinois General Use* 35 Ill. Adm. Code 302.206	Secondary contact and indigenous aquatic life* 302.405	Federal warmwater criteria**, ***
Dissolved Oxygen mg/L All minima should be considered as instantaneous minima to be achieved at all times .	Dissolved oxygen shall not be less than 6.0 mg/L at least 16 hours at any 24 hour period, nor less than 5.0 mg/L at any time.	Dissolved oxygen shall not be less than 4.0 mg/L at any time, 3 mg/L for Cal Sag Channel.	<u>Early life stages present:</u> Lowest 7 day mean 6.0 mg/L 1 day minimum 5.0 mg/L <u>Other life stages</u> 30 day mean 5.5 mg/L 7 day mean minimum 4.0 mg/L 1 day minimum 3.0 mg/L

* Illinois Pollution Control Board, Title 35
** US EPA (1986)
*** The mean and minima are estimated from consecutive measurements of daily average and minimal DO concentrations. The lowest 7 day mean is calculated as the lowest mean of the 7 consecutive daily means while the 7 day mean minimum is calculated as the mean of the lowest 7 consecutive average DO concentrations.

Regarding the formulation of the *DO standard for early life forms present or absent*, the following facts are considered:

1. The USEPA (1986) criteria document specifies that the criteria (standards) for early life stages are intended to apply only where and when these stages occur. The UAA must establish whether the early life stages are present during the time when the lowest dissolved oxygen concentrations occur. The early form designation applies to all embryonic and larval stages and all juvenile forms to 30-days following hatching. The modified impounded warm

water body use such as proposed for the Brandon Pool assumes that physical impairments cause a habitat deficiency that makes it is unsuitable or restricted for development of early life forms. Presence and propagation of early life forms (spawning and hatching) is a necessary condition for a balanced aquatic life. Since the Clean Water Act calls for preservation of biotic integrity and the biotic integrity implies a balanced biota indigenous to the ecoregion, presence of early life forms of tolerant and often foreign species does not mean that the water body can be classified as *early life forms present.*

2. Reference unimpacted streams may exhibit dissolved oxygen concentrations that are below the standard. Unimpacted streams draining wetlands are typically dystrophic and during warm periods have naturally low DOs. Impounding the river for navigation reduces reaeration.

Magnitude. The Illinois General Use standards are similar to the early version of the USEPA warm water quality criteria. The habitat condition and character of the water body make the consideration of cold water standards unrealistic for streams located in the ecoregion.

Considerations were given to the following wording of the USEPA (1986) criteria document

- .... Where natural conditions alone create dissolved oxygen concentrations less than 110% of the applicable criteria means or minima or both, the minimum acceptable concentrations is set at 90% of the natural concentration.... Absolutely no anthropogenic dissolved oxygen depression of the potentially lethal area below the 1-day minimum should be allowed unless special care is taken to ascertain the tolerance of resident species to low dissolved oxygen.
- The USEPA document also states that during periodic cycles of DO concentrations, minima lower than acceptable constant exposure are tolerable so long as:
  - *the average properly calculated concentration attained meets or exceeds the criterion;*
  - *the minima are not unduly stressful and clearly are not lethal.*

This wording allows consideration of daily mean instead of instantaneous minimum for waters that are affected by photosynthetic oxygen production and algal respiration. This contradicts the wording of the DO criterion in Table 8.2. As a matter of fact there has been a considerable and unresolved discussion among USEPA water quality standards specialists as to whether the daily minimum DO concentration is to be applied to an instantaneous minimum or lowest mean daily concentration[4]. The State of Illinois has chosen instantaneous minimum and the US EPA has accepted this interpretation. Table 8.3 presents the levels of protection for warmwater fish species taken from the USEPA (1986) criteria document.

The 1986 criteria document also recommends that if the DO in the water body can be manipulated (e.g., by side aeration) such manipulation could result in extended stress on the aquatic biota by prolonged DO concentrations at or slightly above the DO standard. Because of this effect, the guideline document recommends that the occurrence of the daily minima below the acceptable 7 day mean minimum be limited to 3 weeks per year or that the acceptable one - day minimum be

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[4] Personal communication by Charles Delos (USEPA) to Vladimir Novotny

increased to 3.5 mg/L for warmwater fish. These limiting criteria levels are supported by the literature data that will be presented in the next section.

Table 8.3 Dissolved Oxygen Concentrations (mg/L) vs. Quantitative Level of Effect.

Warmwater (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 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 mortality	3
c	*Invertebrates*	
	No production impairment	8
	Some production impairment	5
	Acute Mortality Limit	4

Literature Review of DO Impacts on Potential Fish Community in the Des Plaines River and Upper Illinois River

Table 8.4 summarizes the potential fish community that could inhabit the Lower Des Plaines and the Upper Illinois Rivers region and Table 8.5 summarizes a series of literature values for DO impacts on the fish species listed in Table 8.4. Using the values in Table 8.5, the DO impact index was developed to translate the narrative impacts into a numerical value. Table 8.6 summarizes the index values used and Figure 8.14 contains a plot of the DO impact index. In accordance with the concepts of the Criterion Minimum Concentration (CMC) and Criterion Continuous Concentration (CCC) defined by the USEPA for priority pollutants, Figure 8.14 shows that to prevent lethal and extremely stressful effects, CMC concentration must be kept above 3 mg/L and to prevent chronic effects, such as distress and growth retardation, CCC DO levels need to be maintained above 4 to 5 mg/l. These CMC and CCC limits would protect all species indigenous to warmwater bodies in the Northeast Illinois ecoregional system. These levels may not provide full protection of large mouth bass. This, limitation, however, is in accordance with the warm- water USEPA (1986a) standing DO criteria.

It should be noted that the literature values in Table 8.5 represent a laboratory sampling conducted between temperatures of 13 and 25°C. Summer temperature values in the Lower Des Plaines River, specifically in the Dresden Island Pool typically exceed these values by as much as 12°C[5]. Under higher temperatures, respiration of fish increases, resulting in higher DO requirements. Therefore, to prevent lethal conditions and provide a margin of safety, it is recommended that 24-hour average dissolved oxygen levels do not drop below 4 mg/L.

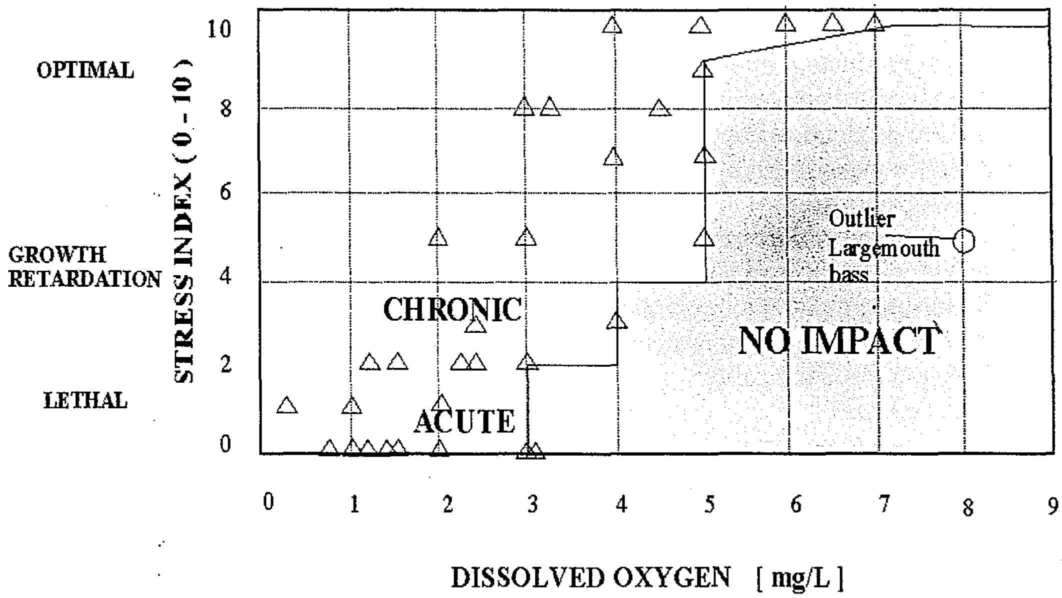

Figure 8.14 Impact of low DO concentrations on fish.
The points represent impacts on the fish indigenous to the Des Plaines River and Upper Illinois River

[5] See Chapter 2 Water Body Assessment - Temperature

TABLE 8.4 Potential Fish Community in the Lower Des Plaines River and Northeast Illinois Rivers Based on Available Habitat Suitability Indexes (HIS) models developed by the US Fish and Wildlife Service

Fish	Current Relative Abundance
Bass – Largemouth	Occasional
Bass – Smallmouth	Occasional
Bass – White	Uncommon
Black Bullheads	Uncommon
Black Crappie	Rare
Bluegill	Common
Buffalo – Smallmouth	Uncommon
Carp	Abundant
Channel Catfish	Occasional
Common Shiner	Record of occurrence available, not sampled
Creek Chub	Probably a stray from a tributary, Lake *Michigan, or inland stocking*
Gizzard Shad	Abundant
Longnose Dace	Not observed
Northern Pike	Rare
Yellow Perch	Rare
White Sucker	Occasionally
Walleye	Rare
White Crappie	Rare

Source: Hey and Associates, Dr. Tim Ehlinger, & EA Engineering, Science and Technology, 1995

TABLE 8.5

Dissolved Oxygen Requirements for Potential Fish Species

Bass ñ Largemouth		
Growth Reduction Level	**<8 mg/L**	(Stewart et al., 1967)
Substantial Growth Reduction Level	**<4 mg/L**	(Stewart et al., 1967)
Distress Level	**5 mg/L**	(Katz et al., 1959; Whitmore et al., 1960; Dahlberg et al., 1968; Petit 1973)
Lethal Level	**<1 mg/L**	(Moss and Scott, 1961; Mohler, 1966; Petit, 1973)
Bass ñ Smallmouth		
Optimal Growth Level	**>6 ppm**	(Bulkley, 1975)
10% Reduced Production Rate	**3 ppm**	(Bulkley, 1975)
20% Reduced Production Rate	**4 ppm**	(Bulkley, 1975)
Lethal Level (20-25°C)	**1 ppm**	(Burdick et al., 1954; Bulkley, 1975)
Bass ñ White		
Lethal Level (21-24°C)	**1 ppm**	(Mount, 1961)
Extremely Stressful Level	**2 ppm**	(Mount, 1961)
Decreased Activity and Coloration	**3 ppm**	(Mount, 1961)
Increased Ventilation	**3 ppm**	(Mount, 1961)
Lower Optimum Limit	**5 ppm**	(Mount, 1961)
Black Bullheads		
Lethal Level (water temp > 18°C)	**3 mg/L**	(Moore, 1942)
Survivable Tension Level – winter	**0.2 ñ 0.3 mg/L**	(Moore, 1942; Cooper and Washburn, 1946)
Optimal Level	**>7 mg/L**	(Carlson et al., 1974)
Black Crappie		
Avoidable Level	**1.5 mg/L**	(Whitmore et al, 1960)
Tolerant Level	**4.5 mg/L**	(Whitmore et al, 1960)
Optimum Growth/Reproduction Level	**>5 mg/L**	(Stroud, 1967; U.S. EPA, 1976)
Lethal Level	**<1.4 mg/L**	(Sigler and Miller, 1963)
Bluegill		
Tolerant Levels – short duration	**<1.0 mg/L**	(Baker, 1941; Cooper and Washburn, 1946; Moss and Scott, 1961; Petrosky and Magnuson, 1973)
Avoidable Levels	**1.5-3.0 mg/L**	(Whitmore et al, 1960)
Optimal Levels	**>5 mg/L**	(Petit, 1973)
Buffalo ñ Bigmouth		
Specific DO requirements are not known; however, **5.0 mg/L** is considered the minimum level for maintaining freshwater fish populations.		(U.S. EPA, 1976)

TABLE 8.5

Dissolved Oxygen Requirements for Potential Fish Species

Buffalo ñ Smallmouth		
Specific DO requirements are not known; however, **5.0 mg/L** is considered the minimum level for maintaining freshwater fish populations.		(U.S. EPA, 1976)
Assumed to be less tolerant of low DO levels than carp which are able to live for short periods at a DO level as low as **3 mg/L**, but optimum DO level **>6 mg/L**.		(Huet, 1970; Jester, personal communication)
Carp		
Adults: tolerant of low DO levels.		
Feeding Levels	**<2 mg/L**	(Hover, 1976)
Elevated Respiration Level (13-23°C)	**3-5 mg/L**	(Itazawa, 1971; Davis, 1975)
Optimum Growth Level	**6-7 mg/L**	(Huet, 1970)
Eggs: tolerant of fluctuating oxygen levels.		
Survival Level – short exposure (25°C)	**1.2 mg/L**	(Kaur and Toor, 1978)
Channel Catfish		
Adequate Growth and Survival Levels	**5 mg/L**	(Andrews et al., 1973; Carlson et al., 1974)
Optimum Growth and Survival Levels	**>7 mg/L**	(Andrews et al., 1973; Carlson et al., 1974)
Growth Retardation Levels	**<3 mg/L**	(Simco and Cross, 1966)
Reduced Feeding Levels	**<5 mg/L**	(Randolph and Clemens, 1976)
Creek Chub		
Specific DO requirements are not known; however, if oxygen requirements are similar to those for other coolwater fishes, concentrations greater to or equal to **5 mg/L** should be sufficient for long-term growth and survival.		(Davis, 1975)
Gizzard Shad		
Minimal Level – absent in water	**2 mg/L**	(Gebhart and Summerfelt, 1978)
Longnose Dace		
Northern Pike		
Short-term Tolerant Level	**0.1-0.4 mg/L**	(Cooper and Washburn, 1949; Magnuson and Karlen, 1970; Petrosky and Magnuson, 1973)
Partial or Complete Winterkill	**<1.0 mg/L**	(Johnson and Moyle, 1969; Stewart, 1978)
Lethal Level (28°C)	**<1.5 mg/L**	(Casselman, 1978)
Yellow Perch		
Winter Lethal Level	**0.2-1.5 mg/L**	(Moore, 1942; Cooper and Washburn, 1949; Magnuson and Karlen, 1970)
Summer Lethal Level (26°C)	**<3.1 mg/L**	(Moore, 1942)
Lower Optimum Limit	**5 mg/L**	(No source listed)
White Sucker		
Avoidable Levels	**< or = 2.4 mg/L**	(Dence, 1948)

TABLE 8.5

Dissolved Oxygen Requirements for Potential Fish Species

Lethal Embryos Level	**< or = 1.2 mg/L**	(Siefert and Spoor, 1974)
Fry Reduced Growth Level	**<2.5 mg/L**	(Siefert and Spoor, 1974)
Walleye		
Adult:		
Short-term Tolerant Level	**2 mg/L**	(Scherer, 1971)
Most Abundant Level	**3-5 mg/L**	(Dendy, 1948)
Lethal Level	**<1 mg/L**	(Scherer, 1971)
White Crappie		
Tolerable Level	**3.3 mg/L**	(Grinstead, 1969)
Lower Limit for Optimal Growth and Survival	**5.0 mg/L**	(Stroud, 1967; EPA, 1976)

TABLE 8.6
Dissolved Oxygen Impact Index

Condition	Index
Optimal Growth Level	10
Decreased Activity and Coloration	8
Increased Ventilation	7
Elevated Respiration Level (13 - 23° C)	6
Growth Retardation Levels	5
Distress Level	4
Substantial Growth Reduction Level	3
Extremely Stressful Level	1
Lethal Level	0

Source: Hey and Associates, Inc.

Ohio DO Standard for the Modified Warm Water Use

The State of Ohio developed and received approval from the USEPA for the Modified Warmwater Use designation. It was pointed out previously in this report that a UAA must be performed before a water body is classified in this category. There is no blank assignment of this modified use to any water body. The magnitude of the DO standard applicable to that use are given in Table 8.7 below.

Table 8.7 Ohio Modified Warmwater Do Standards

Outside Mixing Zone Average	4.0 mg/L
Outside Mixing Zone Minimum	3.0 mg/L

Duration or Averaging of the Minimum Permissible CMC and CCC Concentrations

It has become apparent that USEPA warmwater use with early life forms absent, Ohio warmwater use, and, to some degree, the Illinois current indigenous aquatic life use and secondary contact, have similar magnitudes. However, there is a difference between the duration of the limit, or, duration of the allowed excursion. The Illinois limit of 4 mg/L for indigenous aquatic life use is an absolute instantaneous minimum. The USEPA limit of 4 mg/L is a minimum 7 day mean of daily minima of DO concentrations, and Ohio 4 mg/L is a minimum 24 hour average.

It can be argued, based on Figure 8.14, that 4 mg/L DO standard taken as a daily average provides adequate protection for chronic low DO effects lasting 24 hours or more and 3 mg/L DO standard provides adequate protection for acute (instantaneous) effects of low DO. However, these lower DO levels may not provide adequate conditions for well being of fish population. Hence, the 5 mg/L limit is more appropriate. Using 5 mg/L as an instantaneous limit (Illinois duration of the standard) may be overprotective and, as it was documented in Chapter 2, it may not be attainable for many Illinois streams, even those considered as reference streams. It is recommended that for the modified Brandon Pool use the State of Illinois adopts Ohio's interpretation of the duration, i.e., minimum 24 hour mean DO being at or above 4 mg/L and the absolute minimum being 3 mg/L, which is more protective than the US EPA recommended criterion for early life forms absent situations.

The 5 mg/L, 7 day average of minimal DO concentrations limit for the modified Brandon Pool use may be redundant. Table 8.8 shows a relationship between the minimum daily average and minimum 7 day average concentration for the Lower Des Plaines River during critical periods. It can be seen that the CMC limit of minimum 24 hour average DO of 4 mg/L also provides 7 day average CCC protection at about a 5 mg/L level.

Table 8.8 Minimum Daily and 7 Days mean DO Concentrations

Site	Date	Minimum daily mean DO mg/L	Date	Minimum mean 7 day DO mg/L	Agency
Joliet	8/13-00	4.0	8/10-8/17-00	5.0	MWRDGC
Joliet	7/5-00	4.0	6/10-7/6 - 00	5.1	MWRDGC
I-55	7/31-00	5.6	7/28-8/4-00	6.2	MWG
I-55	6/11-00	5.5	6/10-6/16-99	6.7	MWG
I-55	8/25-98	5.5	8/23-8/29-98	5.9	MWG
I-55	8/4-97	5.4	8/3-8/9-97	6.5	MWG

MWRDGC - Metropolitan Water Reclamation District of Greater Chicago
MWG - Midwest Generation EME, LLC

The daily minimum limit is needed and makes sense in situations where the DO exhibits significant daily fluctuations caused by algal photosynthesis and respiration due to nutrient enrichment, which is the case of the Lower Des Plaines River (see Chapter 2).

There are also differences of the duration definition between the Illinois General Use DO standard and the 1986 federal criterion. The lowest 5 mg/L limit in the Illinois General Use standard and the federal EPA criteria for early life stages present is an absolute minium and, if taken literally, should apply to any measured value, be it a grab sample or the smallest hourly measurement in a continuous program. Chapter 2 has documented that in a nutrient enriched stream significant daily fluctuation of DO concentrations can occur during summer. Under these conditions the instantaneous minimum can drop below 5 mg/L while the daily average is significantly above the 5 mg/L standard. The USEPA (1986) criteria document points out that the DO 5 mg/L limitation could be applied to daily mean values for water bodies where DO concentrations exhibit regular daily fluctuations resulting from nutrient enrichment - photosynthetic effects.

This UAA recommends that the DO standard for the Dresden island Pool is 5 mg/L measured as a daily mean rather than instantaneous minimum. A consideration could also be given to adapting an absolute instantaneous minimum of 4 mg/L.

Formulation of the Proposed Dissolved Oxygen and Other Standards for the Modified Impounded Brandon Road Pool

Assumption and Water Body Characterization

The general use of the water body is not an existing use and the cause of the non-attainment of the use is an existing physical modification of the water body and its habitat that prevents spawning and propagation of early life forms. The DO standard for general use is not attained in the Brandon Road Pool.

The assignment of the general use would be disallowed if

1. The general use is the existing use (e.g., the general use DO and other standard are attained); or
2. The general use can be attained by application of CWA Section 301, 302 and 306 technology based effluent controls of point sources and application of economically feasible and implementable best management practices to nonpoint sources (i.e., the water body is not water quality limited).

The assignment of a use, other than general use, is based on a Use Attainability Analysis prepared for the water body.

Proposed DO Standard for the Modified Impounded Warmwater Body Use (Brandon Pool)

Magnitude and duration:

Minimum daily mean not to be below	4 mg/L
Daily minimum	3 mg/L

The IEPA should also consider developing a frequency of allowable excursions. Currently, the DO concentration is allowed to be less than the standard at flows less than the 7Q 10. Because there is a distinct probability that low DO concentrations may occur more frequently at flows higher than 7 Q 10, as documented in Chapter 2, the frequency component of the standard could be expressed in terms of probability of compliance (e.g., 99.8 percent) rather than an absolute minimum. However, the agency realized that at this time implementation of the frequency component may be legally difficult and we suggested in Chapter 2 that 99.8 percent compliance may, in legal terms, be equivalent to the "no excursion limit".

Ammonium

The Illinois Water Quality Standard (WQS)distinguishes between limits for salmonid fish present (cold water bodies) and salmonid fish absent (warmwater bodies). Similarly to DO standards, criteria for ammonium are divided into those for water bodies with early life forms present or absent. The early life forms absent requires the water body to be classified as a modified impounded warm water body.

Acute WQS

The minimum concentration of total ammonia nitrogen ($NH_4^+ + NH_3$ in mg N/L) does not exceed, the acute limit calculated by the following equation

Salmonid fish are absent

$$CMC = \frac{0.411}{1+10^{7.204-pH}} + \frac{58.4}{1+10^{pH-7.204}}$$

Salmonid fish species are not indigenous to the Des Plaines River/Upper Illinois River System and other warm water bodies. The coldwater classification implicitly implies presence and protection of salmonid fish (USEPA, 1986a) while warmwater classification implies salmonid fish generally absent or not typical for such water bodies. Therefore, the criterion for salmonid fish absent will be used for this modified warmwater body use.

Chronic WQS

The thirty-day average concentration of total ammonia nitrogen (in mg N/L) does not exceed the chronic WQS calculated by the following equation

Early life stages are present (General Use)

$$Chronic\ WQC = \left(\frac{0.0577}{1+10^{7.688-pH}} + \frac{2.487}{1+10^{pH-7.688}}\right) x\ MIN\left[2.85, 1.45x10^{0.028(25-T)}\right]$$

Early life stages are absent (Modified Use for Brandon Pool)

$$Chhronic\ WQS = \left(\frac{0.0577}{1+10^{7.688-pH}} + \frac{2.487}{1+10^{pH-7.688}}\right) x\left[1.45x10^{0.028(25-MAX(T,7))}\right]$$

The Illinois standard of early life stages absent is applied to the Brandon Road Pool during the entire year. The General Use WQS (early life stages present) was adopted by the Illinois Water Quality Board to the Dresden Pool for the period March to October and the early life stages absent for the period November to February.

The new Illinois WQS for ammonium also includes a 4 day average (similar to the priority pollutant criteria):

The highest four day average within the 30-day period should not exceed 2.5 times the CCC.

Other Standards

With the exception of DO and ammonium standards, that in the criteria documents have been linked to presence and absence of conditions for early life forms development, other chemically specific standards will be based on the Illinois General Use standards.

Ohio's Water Use Designation Rule 3745-127 specified that if the biological standard and habitat evaluation demonstrated that the modified warmwater use can be designated for a water body the following situations may occur:

Situation I

- If it is demonstrated that one or more chemical specific or whole-effluent criteria are exceeded, the Ohio EPA or the dischargers can develop and ask for approval a of site specific criterion. Such criterion should be based on the USEPA's *Water Quality Standard Handbook* (i.e. USEPA, 1994), and/or
- Water quality based discharge (effluent) limitations can be developed consistent with the attainment of the designated use.

Situation I may lead to a 303(d) listing and TMDL.

Situation II

- Demonstrated nonattainment of the applicable biological criteria with concomitant evidence that the associated chemical-specific aquatic life criteria and whole-effluent toxicity *are met* will cause the director to seek and establish, if possible, the cause of the nonattainment of the designated use. If the designated use is not attainable the agency will propose to change the designated use. Where the designated use is attainable and the cause of the nonattainment has been established, the director (agency) shall, wherever necessary and appropriate, implement regulatory controls or make other recommendations regarding resource management to restore the designated use. Additional regulatory controls shall not be imposed on point sources that are meeting all applicable chemical specific and whole - effluent criteria unless:

- The point sources are shown to be the primary contributing cause of the nonattainment (e.g., the effluent flow fluctuation and surges are the cause);
- The application of additional or alternative treatment or technology can reasonably be expected to lead to attainment of the designated use.

In Situation II, other means of water quality management can be proposed and employed, including as an ultimate but very effective measure, removal of the impoundment. Other means of water quality management and remediation include in-stream and side-stream aeration, turbine aeration, sediment capping and remediation, dredging of sediments, fish stocking, and others.

Chapter 2 on Water Body Assessment documented that these remaining chemical specific standards are currently met in the Brandon Pool[6]; therefore, the general use defined by these standards is the existing use.

Narrative Standards

In the narrative standard category it is recommended that the State substitutes US EPA wording for "natural origin" (see table 2.2). The Lower Des Plaines River is not a natural river and more than 90 percent of constituents in the river originate from treated effluents and urban runoff. These constituents become "pollution" only when they cause a nuisance or are objectionable..

Effect of the Modified Use Classification on Recreation

Chapter 7 discussed the implications of physical modification of the streams on recreation. While the modified impounded water use designation of the Brandon Pool has some similarities such as an effect on the type of recreation during navigation, in general, impoundments in many cases provide best opportunities for primary and secondary recreation (see Figure 8.4). On the other hand, channels modified purely for navigation with fencing and steep manmade embankments that restrict habitat and recreation of the Brandon Road Pool make it unsuitable for both types of recreation. The linkage and similarities between the modified use designation and limitations on recreation are coincidental.

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[6]The chronic General Use standard for zinc is not met and may not be attainable, while the corresponding federal chronic criterion is attained. These two limits should be reconciled by the agencies before any conclusion on the attainment of the chronic zinc standard is made..

Chapter 7 presented the alternatives for designation of recreation uses for the Lower Des Plaines River and recommended the following use for the Brandon Pool:

<u>Secondary Non-contact Recreation</u>

Because the physical irreversible attributes, navigtation and effluent domination, primary contact recreation is not proposed and is discouraged. However, recognizing the fact that recreation boat traffic through the Brandon Pool is occurring, and the boat launch will be built, the designated use of the pool would be secondary noncontact recreation. The risk for such use should be higher than the risk for primary contact recreation that was recommended between 8 to 14 illnesses/1000 swimmers. This UAA proposes to establish a standard that would recognize the fact that primary contact is either not existent or would be very rare and incidental. This standard would be five times 548 cfu of E. Coli/100 mL which is five times the criterion based the highest primary contact risk of 14 illnesses/1000 swimmers[7] . The standard is then 2740 cfu/100 mL of *Escherichia Coli* indicator organisms measured as geometric mean of samples. No single maximum standard is proposed.

This water quality, expressed by the fecal coliform densities, is existing, i.e., the currently measured geometric mean of 350 fecal coliform bacteria cfu/ 100 mL is greatly below the proposed secondary use standard of *E. coli*. The AquaNova/Hey Associates team feels that, in the next standard evaluation cycle, the agency could adopt a standard that would be based on a smaller risk. For example, the water body could meet a secondary standard based on the value five times the lowest risk (8 illnesses/1000 swimmers) that is 630 EC cfu/1000 mL; however, the difference between the proposed standard and current geometric mean provides a margin of safety. Because the *E.coli* densities must be less than that of fecal coliforms (*E. coli* is a part of fecal coliform group) it can be stated with a great scientific certainty that the current water quality would meet the proposed *E.coli* geometric mean standard for secondary recreation and water quality at this level is existing.

Pathway for Determining the Modified Impounded Warmwater Use

The key decision points of the eligibility of the water body to be classified as a modified warmwater use are:

1. *The water body has been irreversibly (in the long run) physically modified by humans by impounding the river for existing beneficial purposes such as navigation and removing these uses would cause a widespread socio-economic impact .*
2. *The water body is not meeting the general use biotic criteria but is meeting or could meet the modified impounded biotic standards.*

Fish composition is especially important if it indicates that early life forms of a balanced fish assemblage are not present or are present in small numbers but do not originate from the site (e.g., they may be passing trough and/or their presence is accidental).

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[7]At the conclusion of this study we were informed by the representative of the USEPA, Region V, that the acceptable maximum risk may be reduced to 10 illnesses per 1000 swimmers.

The indices of biotic integrity and sediment contamination express and represent a long term impact of stresses on water quality and aquatic less tolerant species. Sediment contamination may be caused by legacy pollution that occurred years ago. Poor biotic evaluation reflects the impact of stresses dating a few years in the past while chemical evaluation reflect the water quality at the time when the sample was taken. Unless high frequency or continuous sampling is used biotic and sediment evaluation is more reliable. Therefore, isolated (outlier) violation of a chemical based water quality standard while biotic evaluation indicate attainment may not be a reason for classifying the water body in this modified use category.

If the impairment is caused primarily by excessive waste loads from point and nonpoint sources the water body is not eligible for the modified use designation unless the Loading Capacity of the water body was significantly reduced by the physical man-made features of the water body.

Evaluation and Use Designation of the Dresden Pool

Based on the evaluation of the existing water quality, habitat, attainable water quality and biotic assessment it is recommended that

the General (Impounded) Use designation is extended to the entire Dresden Island Pool

The standards applicable to the Dresden island Pool will be Illinois General Use Standards. Site specific standards are recommended for copper and dissolved oxygen. The "impounded" subuse designation recognizes the fact the level of biotic integrity of impounded waters is not commensurate with the biotic integrity values typical for wadeable streams (see Chapters 5 and 6 for biotic integrity assessment based on criteria developed for smaller wadeable streams).

For temperature, this UAA has found that the current Secondary Contact and Indigenous Life Standard does not protect the aquatic life (fish) from lethal effects of temperature and recommended that the temperature standard for the General Use is used. The UAA also concluded that the first five reasons for the change of the use or standards (Box 1.1) cannot be used to modify the General Use Standard to provide relief to the dischargers of heated effluents to the Dresden Island Pool. In view of the anticipated expense for the installing cooling at the Midwest Generation Joliet power plants needed to meet the general use WQS it is expected that the Agency will give an opportunity to the Midwest Generation and the stakeholders to prepare a socio-economic study documenting that the cost associated with meeting the general use standard would result in a substantial and wide spread impact. Cost alone cannot be used to justify a downgrade from the General Use temperature standard unless this cost represent a substantial and wide spread adverse impact on the dischargers and population. Installing cooling technology is common at many thermal power plants without causing a substantial and wide spread socio-economic impact. Virtually no other state has temperature standards higher than the Illinois General Use temperature standard, even for "marginal" waters.

Primary recreation use and the uniform standard for pathogens are recommended to be extended to the Dresden Island Pool as follows:

The Upper Dresden Island pool has natural assets that promote primary recreation, especially in the section downstream of the river mile 283. However, this stretch of the river also has a relatively high concentration of industrial activities and most recreation will still occur downstream of the I-55 bridge. The expected frequency of swimming will be still low and frequency of the primary contact recreation will be much less than in the other Illinois streams. Therefore, the state may choose a higher acceptable risk of 14 illnesses/1000 swimmers (see Footnote 7). It is also expected that the frequency of the primary use would be characterized as "Infrequently Used Full Body Contact" or as "Marginal Primary Contact Recreation".

The *E.coli* based standard for this level of risk would then be (Table 7.1):

Geometric mean density of *E.coli* 548 cfu/100 mL

The single value maximum is for beach closings and swimming advisories:

From Table 7.1 the maximum value corresponding to the risk of 14 illnesses per 1000 swimmers is 2507 E.Coli cfu/1000 swimmers.

Using enterococci as indicator organisms is not recommended because they are primarily used for marine beaches.

The IEPA and the Illinois Pollution Control Board may chose to adopt a lower risk of contacting waterborne illness; this is up to the state discretion.

The FC based standard should be discontinued. Due to the fact that there is a great similarity between the *E. Coli* and fecal coliform densities and *E.coli* density cannot exceed that of fecal coliforms, continuation of the fecal coliform based standard does not make sense. In the next year, the agencies and dischargers should focus on developing data bases for E/Coli indicators.

The proposed standards are attainable (with disinfection of Joliet effluents) and would provide adequate protection for contact recreation in the entire Dresden Island pool.

- Abandon the maximum limit of 10% of samples can exceed 400 FC cfu/100 mL that is not attainable in the Lower Des Plaines River and its reference sites and is overprotective based on recent USEPA (2002) draft standard guidelines.

Copper. The WQS for copper can be adjusted by developing a Water Effect Ratio that would relate the copper toxicity obtained in the laboratory to that in the river. The river water contains many ligands that detoxify the metal that were not present in the laboratory water of the bioassay experiments from which the copper standard was derived. The methodology for the WER estimation is described in the Water Quality Standard Handbook (USEPA, 1994).

Zinc. Responsible agencies should reconcile the large difference between the "new" Illinois chronic standard for zinc and corresponding federal criterion. After the reconciliation the question of attainment and attainability should be revisited. It ios our opinion that the Illinois chronic standar is overprotective and unattainable.

Conclusions

The Brandon Pool is classified by the proposed Brandon Modified Impounded Warmwater Use Designation

Definition and assignment of the modified impounded warmwater use for the Brandon Pool will not lead to a blanket relaxation of the chemically specific standards below those for the General Use standards. One of the main objectives of designating the modified impounded warmwater use is to recognize the fact that habitat and conditions for a balanced aquatic biota are irretrievably affected and cannot be remedied. If the physical cause is reversible and can be remedied the assignment of the modified impounded warmwater use will lead to a realistic water body restoration.

This UAA documents that this special use for the Brandon Pool is appropriate and less stringent standards for dissolved oxygen, ammonia and Escherichia Coli can be applied. The rest of the chemical WQS are derived from the Illinois General Use standards, including those for copper and temperature. Reevaluation of the new General use chronic standard for zinc and its reconciliation with the corresponding federal criterion is recommended. Because the General Use standards are attained[6] or can be attained by application of CWA Section 301, 302, and 306 technology based effluent controls of point sources and by economically feasible and implementable best management practices for nonpoint sources, the Illinois General Use standards shall be applied.

The General (Modified) Use designation should be extended to the entire Dresden Island Pool with the associated standards. Site specific standards should be applied for dissolved oxygen, copper, and Escherichia Coli.

Although the portion of Dresden Island pool studied and evaluated by this UAA extends only from the Brandon Road Dam to I-55 bridge, unifying the standard for the entire pool to the confluence with the Kankakee River makes sense and will not affect the current General Use standards applicable to the reach from I-55 to the Kankakee River. This UAA also recommends that regarding temperature, the General Use thermal standard is necessary and appropriate to protect the aquatic community otherwise attainable within the Upper Dresden Island pool. However, economic and operational considerations may be significant and should be given due consideration in the development of any alternate standards and the compliance period to attain that new standard. The Agency should work closely with Midwest Generations and other affected thermal sources to accurately estimate the technical, financial and scheduling requirements. If attainment of the Illinois General Use Standard is found to cause a substantial and wide spread socio - economic impact, we recommend that a new standard include a maximum temperature that represents the upper bound to prevent lethality of known indigenous fish species and additional criteria to address general growth and health needs of aquatic life effects.

Summary of Standards for Brandon Road Pool

DO Standard for the Modified Impounded Warmwater Body Use

Magnitude and duration:

Minimum daily mean not to be below	4 mg/L
Daily instantaneous minimum	3 mg/L

Ammonium

Acute criterion

The one hour average concentration of total ammonia nitrogen ($NH_4^+ + NH_3$ in mg N/L) does not exceed, more than once in three years on average, the CMC calculated by the following equation

Salmonid fish are absent

$$Acute\ WQS = \frac{0.411}{1+10^{7.204-pH}} + \frac{58.4}{1+10^{pH-7.204}}$$

Chronic criterion

The thirty-day average concentration of total ammonia nitrogen (in mg N/L) does not exceed, more than once every three years on average, the CCC calculated using the following equation

Early life stages absent (entire year)

$$Chronic\ WQS = \left(\frac{0.0577}{1+10^{7.688-pH}} + \frac{2.487}{1+10^{pH-7.688}}\right) x \left[1.45 x 10^{0.028(25-MAX(T,7))}\right]$$

where T is the temperature in °C.

The highest four day average within the 30-day period should not exceed 2.5 times the CCC.

Bacteria

The standard for the secondary contact is 2740 cfu/100 mL of *Escherichia Coli* indicator organisms measured as geometric mean of samples. No single maximum standard is proposed.

Remaining standards for the Brandon Pool are the Illinois General Use standards given in the summary table below.

Numeric General Use Standards

Parameter	Illinois General Use Standards Title 35:Env. Protection, C:Wat.Pollution, CH. 1	
pH (units = log [H^+])	6.5 - 9	
Toxic compounds	Acute	Chronic
Arsenic (µg/l) trivalent-dissolved	360*1.0	190*1.0
Cadmium (dissolved)[1)] (µg/l)	exp[A+Bln(H)]x {1.138672-[(lnH)(0.041838]}* A=--2.918 B=1.128	exp[A+Bln(H)]x {1.101672-[(lnH)(0.041838]}* A= -3.490 B= 0.7852
Chromium (total hexavalent) (µg/l)	16	11
Chromium (trivalent-dissolved)[1)] (µg/l)	exp[A+Bln(H)]x 0.316* A= 3.688 B=0.819	exp[A+Bln(H)]x 0.860* A=1.561 B=0.819
Copper (dissolved)[1)] (µg/l)	exp[A+Bln(H)]x 0.96* A= -1.464 B=0.9422	exp[A+Bln(H)]x 0.96* A= -1.465 B=0.8545
Cyanide (µg/l) total	22	5.2
Lead (dissolvedl)[1)] (µg/l)	exp[A+Bln(H)]x {1.46203-ln(H)(0.1457120]}* A= -1.301 B=1.273	exp[A+Bln(H)]x {1.46203-[(lnH)(0.145712)]}* A=-2.863 B=1.273
Mercury (dissolved) (µg/l)	2.6x0.85*=2.2	1.3x0.85=1.1*
Nickel (dissolved)[1)] (µg/l)	exp[A+Bln(H)]x 0.998* A=0.5173 B=0.8460	exp[A+Bln(H)]x 0.997* A=-2.286 B=0.8460
TRC (µg/l)	19	11
Zinc (dissolved) (µg/l)	exp[A+Bln(H)]x 0.978* A=0.8875 B=0.8473	exp[A+Bln(H)]x 0.986* A=0.8604 B=0.8473 Adoption of federal criterion is recommended
Benzene (µg/l)	4200	860
Ethylbenzene (µg/l)	150	14
Toluene (µg/l)	2000	600
Xylene (µg/l)	920	360

Footnotes (March 2001 Draft)

ln[H] is a natural logarithm of hardness
*Conversion factor (translator) for dissolved metals

<u>Conversion factor</u> means the percent of the total recoverable metal found as dissolved metal in the toxicity tests to derive water quality standards. These values are listed as components of the dissolved metals water quality standards to convert the total metals water quality to dissolved standards and were obtained from the USEPA water quality criteria. In the federal criteria this parameter is represented by the Water Effect Ratio.

<u>Metals translator</u> means the fraction of total metal in the effluent or downstream water that is dissolved. The reasons for using a metals translator is to allow the calculation of total metal permit limits from a dissolved metal water quality standard. In the absence of site specific data for the effluent or receiving water body, the metals translator is the reciprocal of the conversion factor. **If dissolved metal concentrations are used, the underlined conversion factor (translator) needs to be used when dissolved concentrations are compared to the standard. The translator needs not to be used when total concentrations are compared to a standard.**

Table 8.9 - Continued

Parameter	Illinois General Use Standards
Barium (total) (mg/l)	5.0
Boron (total) (mg/l)	1.0
Chloride (mg/l)	500
Fluoride (mg/l)	1.4
Iron (dissolved) (mg/l)	1.0
Manganese (total)(mg/l)	1.0
Phenols (mg/l)	0.1
Selenium (total) (mg/l)	1.0
Silver (total)[1] (μg/l)	5.0
Sulfate (mg/l)	500
Total Dissolved Solids (mg/l)	1000
Temperature	32°C (Apr.-Nov.) 16°C (Dec. - March)[3]
Radioactivity Gross beta (pCi/l) Radium 226 (pCi/l) Strontium 90 (pCi/l)	 100 1 2

Table 8.10 Comparison of Narrative Illinois State General Use And Secondary Contact And Indigenous Aquatic Life Use Standards With Federal Aquatic Life Protection And Water Contact Use Criteria

Parameter	Illinois General Use Standards	Federal Aquatic life and Human Health Protection Criteria
Narrative Objectionable floatables	~~(Waters of the state shall be free from sludge or bottom deposits, floating debris, visible oil, odor, plant or algal growth, color or turbidity of other than natural origin).~~ **It is recommended that Federal Aquatic Life Criteria wording is accepted for the Lower Des Plaines River due the fact that the flow of the river is not ìnaturalî.**	All waters free from substances attributable to wastewater or other discharges that: (1) settle to form objectionable deposits; (2) float as debris, scum, oil, or other matter to form nuisances; (3) produce objectionable color, odor, taste, or turbidity; (4) produce undesirable or nuisance aquatic life
Algae		
Odor, color and turbidity		

Footnotes:

1) The limiting concentration for metals is calculated from

$$C = \exp[A + B \ln(H)]$$

where ln[H] is a natural logarithm of hardness

2) The standard of 200 No/100 ml is applied to a geometric mean of a minimum of five samples taken over a 30 day period, the standard of 400 No/100 ml can be exceeded by no more than 10% of samples during any 30 day period.

3) The water temperature should not exceed 32°C (April - November) and 16°C (December-March) during more than 1% of the hours in the 12-month period ending with any month. Moreover, at no time shall the water temperature exceed the maximum limits (32 and 16) by more than 1.7°C.

End of footnotes

Summary of Standards for the Dresden Island Pool

Dissolved Oxygen

This UAA recommends that the DO standard for the Dresden island Pool is 5 mg/L measured as a daily mean rather than instantaneous minimum. Consideration could be given to adopting an instantaneous minimum of 4 mg/L.

Copper

WQS = General Use WQS/WER

where the water effect ratio is ascertained following the methodology included in USEPA (1994) handbook.

Zinc

Considerations should be given to make the Illinois chronic standard at the same level as the federal chronic criterion.

Bacteria

The *E.coli* based standard for the level of risk of 14 ilnesses/1000 swimmers is

Geometric mean density of *E.coli*	548 cfu/100 mL

This use should be characterized as **Marginal Primary Contact Recreation.**

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CHAPTER 9

ACTION PLAN

Introduction

In the preceding chapters we have identified the water quality problems of the Lower Des Plaines River and addressed remedies particular to each problem. It is clear that the Lower Des Plaines River is a highly modified water body that does not resemble its pre-development status. The physical modification and attributes are mostly irreversible in the long-term. However, the main goal of the UAA is to find an ecologically optimal state that would as closely as possible and economically (without causing an adverse wide spread socio-economic impact), approach the goals of the Clean Water Act. We have also stated that the river needs continuous help and should be managed in order to reach these goals.

This UAA has found that the water quality situation of the river has significantly improved since the 1970s when the Secondary Contact and Indigenous Aquatic Life use designation was defined by the Illinois Pollution Control Board. The water and sediment quality today is also better than that measured ten years ago. In 2000, a majority of chemical water quality parameters met the Illinois General Use standards (see Chapter 2). Sediment quality has also improved (Chapter 3). None of the analyzed sediment quality parameters in 1999-2000 by the Illinois EPA and MWRDGC were classified as highly elevated according to the IEPA scale (see Chapter 3). However, sediment contamination by PCBs and several toxic pesticide byproducts in the sediments, revealed in the USEPA 2001 extensive survey, warrant a remedial investigation, especially at the River Miles 286+ and 282.

The conditions in the Lower Des Plaines River have been steadily improving. After the common sense actions outlined in this report and summarized in this chapter are taken the potential for further improvement will increase and the Lower Des Plaines River in the Dresden Island pool could meet the general use classification. This potential for improvement is real; however, the water body may never reach the ecological status of pristine wadeable streams.

In Chapter 8 we have outlined the goals for the Lower Des Plaines River

For the Brandon Road Dam Pool we have developed a new *Modified Impounded Use Designation* and suggested to the Illinois EPA to present it to the Illinois Pollution Control Board and, subsequently, to the US Environmental Protection Agency for approval. This use designation allows adaptation of the dissolved oxygen standard for early life stages for this specific segment as specified by the USEPA (1986) water quality criteria document.

In Chapter 7, we have evaluated the recreational use of the river. Due to the severity and irreversibility (in the long-term) of the physical structure of the channel, and for safety considerations we concluded that the Brandon Road Dam pool was not suitable for primary recreation. We suggest that the Illinois Environmental Protection Agency proposes a secondary contact use for the pool based on the *Escherichia Coli* indicator levels five times the value of the standard suggested for the infrequent primary contact category. This level of protection will allow limited use of the pool for noncontact recreation such as boating, fishing and aesthetic enjoyment of the river and will provide adequate protection for incidental contact with water related to those activities. We have noted that the State of Illinois may also have an option not to provide protection to the recreational users in this segment but recommended not to use this option because the evidence has shown that the bacteriological quality of the Brandon Road poll could meet the secondary use standards. Also the City of Joliet is making a legitimate effort to promote the river and the use of the Brandon Road pool for secondary recreation by building the riverside park and developing a public landing. Providing opportunities for recreation are needed in this large urban community.

Because most of the General Use standards for chemical parameters, including temperature, are met by the existing water quality, the General Use standards should be applied to those parameters that meet or could potentially meet them. The new standards and proposed modifications are included in Chapter 8. The new standards different from the General Use for the Brandon Pool include

Dissolved Oxygen
Bacteria and
Copper
Zinc (chronic)

The modified impounded use classification represents the ecologic potential of the Brandon pool. The modified use of the Brandon pool and the secondary contact uses are subject to periodic reviews certifying to the USEPA that the physical attributes of the pool have not changed. Future periodic recertification may not necessitate a full scale Use Attainability Analysis.

For the Dresden Island Pool we have documented in Chapter 2 that the Illinois General Use, expressed by the mandatory chemical standards, is attained or attainable for the entire pool provided that certain remedial actions are taken which we perceive as not causing a wide spread adverse socio-economic impact. Chapter 4 documented that, unlike the Brandon Road Dam pool where the habitat is severely restricted and constricted, most of the Dresden Island pool has fair to good physical habitat conditions, starting with an excellent but impaired by pollution habitat zone at the confluence of the river with Hickory Creek. The habitat in the Upper Dresden Pool (above I-55) is similar to that in the Lower Dresden pool. Although the current habitat conditions of the Dresden Island Pool do not meet the criteria for habitat assessment developed by the State of Ohio, it may be possible to meet them in the future.

However, the evaluation of biotic integrity using fish IBI (Chapter 6) revealed that due to the impounded character of the river, the Illinois biotic general use guideline value is not attainable. However, there is no mandatory biotic integrity standard. Most impounded streams in this ecoregion do not meet this guideline value. The State of Ohio recognized this problem by instituting a lower IBI criterion for impounded waters commensurate with other impoundments. Therefore, we propose that the Illinois EPA accepts this scientific finding and adopts the reduced biotic integrity status for the Lower Des Plaines River in the Dresden Pool similar to the other impounded streams as the near future ecologic potential.

The Lower Des Plaines River in the Dresden pool is an impounded water body heavily used for navigation and containing legacy pollution in sediments. It was noted and documented (in Chapters 4 to 6) that such water bodies cannot reach an ecological status comparable to the unmodified free flowing streams; however, they can reach a status of a balanced biota indigenous to the impounded water bodies and water quality that meets most or all important chemical and microbiological water quality standards. Because the chemical water quality in the entire investigated Lower Des Plaines River, with exception of the DO (in both pools) and temperature (in the Dresden Island pool) meet the Illinois General Use standards, these standards should be adopted. Therefore, the proposed use for the Lower Des Plaines River is a form of the general use for impounded water bodies and not a special use that would allow an unsubstantiated relaxation of the General Use standards.

The following modifications of the General use standard are proposed for the Dresden Island pool:

DO standard expressed for daily mean and absolute minimum
Copper modified by the Water Effect Ration to be developed for the segment.
Chronic zinc standard at the level of the federal CCC criterion

The study proposes that the temperature standard is made commensurate with the General Use standard. The current Secondary Use and Indigenous Aquatic Life standard for temperature does not provide a protection against the lethal temperature levels.

Parameters and conditions of concern that may have to be addressed in the long run include mercury, nutrients and contaminated sediments. Also, the ecologic potential expressed by the Indices of Biotic Integrity may have to be periodically reassessed. Adequate mercury assessment will require a change to more sensitive "clean" methodologies. Nutrients are very high in the river but, in the absence of standards that would link the nutrient levels to impairment of the integrity, we recommend that the nutrient question be addressed in the future reassessment.

The short- and long-term remedial actions outlined below are in agreement with the Adaptive Management Concept proposed and highlighted by the Committee to Assess the Scientific Basis of TMDL (2001). The short-term actions will have immediate and significant beneficial impacts on the integrity of the two pools. Noting that most chemical parameters in the river already meet the General Use standards, implementing the short-term measures may bring about an attainment of the majority of the goals. Implementation of the long-term measures should be delayed until after the short-term actions have been implemented and assessed, which may require a period of about five years.

Sediment Contamination

Both investigated sections of the Lower Des Plaines River are impounded. However, sediment deposition is limited mostly to areas outside of the navigational channel. In the Brandon Road Pool, a depositional zone is located upstream of the dam because the navigation is diverted to the lock channel. In the Dresden Island pool, sediments can also deposit in the downstream tail water of the Brandon Dam outside of the navigational channel between River Miles 279 and 282.

The sediment contamination is less in the Dresden Island pool and none of the sediment quality parameters measured by the Metropolitan Water Reclamation District of Greater Chicago were more than highly elevated (>98th percentile). The sediment quality for most parameters is between less than 85th and 98th percentile of quality of Illinois riverine sediments which would be classified as elevated according to the IEPA scale.

In 2001, the USEPA conducted a comprehensive and extensive survey of sediments in the Lower Des Plaines River and analyzed three times in this year for many parameters, including conventional sediment composition (TS, VSS, nutrients), metals, asbestos, cyanides and organic pollutants (PCB, PAHs, pesticides, and other organics). We have used sediment partitioning concept to calculate the sediment toxicity units (STUs) for these pollutants as guidance for assessment. This method relates the calculated pore water concentrations to a guidance chronic water only criterion for the substance. In the absence of any specific sediment standards this was the only method to identify pollutants of concern; however, application of the Sediment Toxicity Unit concept in this study has no regulatory implications. Other methods, non binding for assessing the toxicity problems, have been proposed in literature and used elsewhere, for example, by the State of Minnesota or Province of Ontario. The State of Illinois should revisit the problem of identification and ranking of the contaminated sediments.

We found that PCBs and three pesticide residues are potentially problematic. The levels are an order of magnitude worse in the depositional hot spot at River Mile 286+ (upstream of the Brandon Road Dam). Other sediment pollutants such as metals and PAHs are not found at levels of concern.

Proposed Actions

Short-Term Actions

Actions by the Illinois Environmental Protection Agency and Illinois Pollution Control Board

This UAA has reviewed the Illinois General Use Standards and found that standards for some parameters are different from the federal water quality criteria or draft criteria (e.g., USEPA, 1986; USEPA, 1999; and USEPA 2002) and are sometimes overprotective. While extra protection is commendable, it may result in a situation where no action, short of treating the entire river flow, would result in attainment. Unattainable standards are one of the reasons for hundreds of failing TMDLs (Houcks, 1999; Committee to Assess the Scientific Basis of TMDL Program, 2001).

The following revisions of the Illinois General Use Standards are proposed:

1. Adopt the federal criteria for pathogens and establish a secondary contact use for the Brandon Road pool and a primary higher risk recreational use for the Dresden Island pool. Federal criteria recognize acceptable risk between 8 to 13 sickness cases/1000 swimmers to select; however, the low risk (8 sicknesses/1000 swimmers) is appropriate for highly frequented beaches, which is not the case nor is it proposed for the Lower Des Plaines River. This UAA recommended using the highest risk of 13 sicknesses /1000 swimmers for the definition of the geometric mean and maximum concentrations of the Escherichia Coli indicator and abandoning the current Fecal Coliform indicator. The risk level should be periodically reevaluated and the standard adjusted accordingly in the future.

2. For the Lower Des Plaines River only, express the magnitude of the dissolved oxygen standard as a minimum 24 hour mean DO (5 mg/L in the Dresden pool and 4 g/L in the Brandon pool) and absolute minimum (4 mg/L in Dresden pool and 3 mg/L in the Brandon pool). The 7 day mean or minimum may be redundant and unnecessary.

3. Develop a Water Effect Ratio for metals based on toxicity difference between the waters of the Lower Des Plaines River and the laboratory water for which standards were developed in the laboratory.

4. Reconcile the large difference between the General Use chronic standard for zinc and corresponding federal CCC criterion. The General use standard appears to be overprotective.

5. The current temperature standard for the Brandon pool is not protective of the existing and proposed use and should be changed to the General Use standard. However, the dischargers of heated flows and stakeholders should be given the opportunity to address the socio-economic impact of the temperature standard.

Recommendations unrelated to modifications of the standards and use are:

6. Continue biotic monitoring and utilize IBIs for assessing the biotic status of the river relative to current ecological expectations expressed in terms of IBI goals for the impounded Dresden Island pool and modified impounded Brandon Road pool as short-term measures of attainment that will be reassessed later (five years to ten years from the beginning of

implementation of the program) when the effect of short term measures will become evident by monitoring.

7. Continue the chemical monitoring program and improve detection limits for some parameters (e.g., mercury). Begin bacteriological quality monitoring of the state waters using Escherichia Coli as indicator microorganisms.

8. Consider establishing a water quality management system and coordinating group for the Lower Des Plaines River that could be expanded to include the entire Des Plaines River watershed, including Chicago Waterway System (pending completion of the UAA for the Chicago Waterways). This water management system could carry out daily forecasting of water quality levels in the river, issue warnings to swimmers and other recreational uses, issue warnings when toxic spills occur, and operate or advise on operation of aeration at the Lockport Dam and power house, based on forecasted DO emergencies. The river also needs fish management such as restocking with higher quality fish and protection and maintenance of fish spawning grounds.

Actions by the Dischargers and Users of the Brandon Road Dam Pool

The short term actions could be possibly implemented within five to ten years. There are two problems that should be remedied in the short-term in the pool:

1. **The dissolved oxygen concentration**. The DO concentration in the Brandon Road pool does not meet the proposed standard for the modified Brandon pool use or the Illinois Secondary Contact and Indigenous Aquatic Life and General Use standards. It is obvious that meeting the existing or proposed (4 mg/L 24 hour average, 3 mg/L minimum) standard is tied to the actions that will occur upstream of Lockport Lock and Dam in the Chicago Waterway System that is being studied by another Use Attainability Analysis.

   To alleviate and resolve the dissolved oxygen problem in the Brandon pool we suggest that, in the short-term, the MWRDGC considers aeration at the Lockport dam and power house. The DO modeling presented in Chapter 2 has shown that if the DO standard is maintained in the downstream tail water of the Lockport Dam it will be maintained also throughout the Brandon pool. Turbine aeration and aeration over the spillway are very effective in-stream measures to supplement the DO. Turbine aeration which was practiced, for example, in the Ruhr River district in Germany, requires modification of the turbines. There is an unused spillway attached to the Lockport lock over which water can be released by pumping, creating supercritical flow on the spillway that has a very high aeration capacity. Aeration over the Brandon Road Dam is an example and proof of the attainability of the DO downstream of the Lockport Dam. The MWRDGC could also develop a DO forecasting system tied to the continuous DO and temperature measurements in Joliet and effluent and CSO discharges upstream in the Chicago Waterways that would alert the river managers about the possible DO excursion and implement aeration measures at Lockport.

   Long-term DO management is tied to the actions taken and implemented in the upstream Chicago Waterways.

2. **Toxic content of the sediments.** Contamination of the Brandon Road poll sediments is elevated in several depositional sections. Also, sediment contamination by PCBs and several pesticides is high. In the navigational section the bottom sediments are composites of bedrock and gravel and may not be toxic (see Chapter 3 and Burton, 1995) and the sediment contamination therein may not be elevated or be only mildly elevated (with exception of PCBs

that are also high in the sediments of the navigational channels). **The current sediment contamination is not restricting implementation of the General Use classification for the Lower Dresden Island pool.**

The urban areas discharging stormwater into the Lower Des Plaines River and upstream communities should implement nonpoint pollution control programs for reducing toxic and bacteriological pollution of urban runoff. This is a necessary component of the sediment toxicity control program. The current trend in sediment contamination is toward improvement.

Regarding the PCB contamination, we propose that a remediation study be conducted that should be extended to the CSSC. The study should include long term modeling of the fate of the PCBs and of the three pesticides (dieldrin, chlordane, and heptachlor epoxide) in the sediment and in water, considering the effect of navigation, degradation, uptake by algae and convective transport by water. The study should include a comprehensive assessment of the distribution of the conataminats and toxicity of the sediments throughout the area and propose and assess remediation of the hot spots by capping or sediment removal and possible remediation (including recovery by no action) of contaminated sediments in and out of the navigational channels.

3. **Limited use of the Brandon pool for recreation.** The governing bodies should post warnings, maintain railing and fencing along the Brandon pool and conduct public education to prevent use of the pool for swimming, especially by children.

Meeting the DO standard for the modified use hinges on meeting either the current Secondary Contact and Indigenous Aquatic Life Use DO standard or the new modified Brandon pool standard immediately downstream of the Lockport Lock and Dam (a part of the Chicago Waterway System and not a part of this UAA). The subsequent UAA for the Chicago Area Waterways System will address the attainability of the standard at Lockport. If the standard is not attainable upstream of Lockport Lock and Dam, in-stream aeration can be implemented during times when the DO in the Brandon pool would be expected to drop below the DO standard for the Brandon pool. The in-stream aeration by turbine aeration or flow over the spillway may last only a few days during some years and may not constitute a wide spread adverse socio - economic impact.

This UAA is not recommending a water quality TMDL for the Brandon Road Dam pool, provided that the proposed actions are considered and implemented.

The Illinois IEPA should also consider expressing numeric standards in terms in three dimensions, i.e., magnitude and frequency and duration of allowable excursion, or, alternatively, in terms of the probability of compliance. The frequency of allowable excursions is included in water quality regulations (40 CFR 131) as once in three years for biological excursions or during flows that are less than the 7Q10. Similar allowable excursions could be extended to other parameters such as dissolved oxygen, temperature and ammonium. These allowable excursions are very rare and do not diminish water quality. There is also a substantial margin of safety incorporated in the magnitude of the EPA criteria. Adopting the three dimensional standards will allow unbiased statistical water quality assessment (see Chapter 2).

Actions of Dischargers and Users of the Dresden Island Pool

There are three problems that prevent full attainment of the ecologic potential. One is contamination of the sediments by three pesticides (pesticide byproducts) and PCBs. The second problem is the absence of disinfection of the effluents discharging sewage with high levels of bacteria into the Dresden Island pool (primarily Joliet East and West and those plants on Hickory Creek that do not practice disinfection). The third problem is the temperature in the Upper Dresden Island pool.

The Dresden Island pool does not have the habitat impairing physical deficiencies such as those recognized for the Brandon Road pool. The river is impounded and wider than in the Brandon pool. However, the fact that the river is impounded and used heavily for navigation means that the ecologic potential is significantly less than that for free flowing natural rivers. As far as chemical parameters in water are concerned, the Dresden Island pool meets the General Use standards for all parameters except mercury, temperature, and chronic zinc. Attainment of the chronic standard for copper was marginal at MWRDGC sites 94 and 95 that measure total copper and would require implementing toxicity-based WER correction of the Cu standard for the pool. The dissolved oxygen standard would not be met if the general use standard is literally interpreted as not being exceeded at all times. However, in the interpretation of the USEPA criteria the standard would have been met if average daily concentrations had been considered.

Potential Toxicity of the Sediment in the Downstream Tailwater of Brandon Road Dam

Burton's (1995) study identified this sediment as highly toxic. With exception of PCBs (not analyzed by Burton) the current sediment analysis does not confirm the high toxicity levels (Chapter 3), at least not at the level measured by Burton. Nevertheless, because this area has been a receptor of the effluent and CSO from a large urban area, the toxicity problem cannot be discounted and must be addressed. Generally, urban runoff and not domestic sewage is the source of toxic contaminants (USEPA, 1983). Runoff from industrial areas must also be included. These sources are subject to the NPDES stormwater permitting and development of stormwater control programs. It should be also pointed out that the extremely low chronic toxicity standard for PCBs is related more to protection of humans eating fish and drinking contaminated water after PCB biomagnification through the food web than to protection of aquatic life.

Surprisingly, channel sediment contamination measured by the 2001 USEPA study in Brandon Road and Dresden Island pools are similar, indicating that the sediment contamination was almost evenly spread by the navigation impact throughout the entire Lower Des Plaines River.

Recommended Remedial Actions

This UAA recommends that the City of Joliet completes its program of elimination of CSOs and also considers effective best management practices for control of toxicity in the urban runoff. The Nationwide Urban Runoff Project (USEPA, 1983) and many follow up studies have found that urban runoff contains elevated concentrations of metals and Polyaromatic Hydrocarbons (PAHs) that are a source of toxicity. However, the current sediment levels of metals and PAHs may not be acutely toxic and an accurate evaluation of chronic toxicity by these compounds may be difficult for the lack of scientific evidence and criteria. For the key pollutants (PCBs) the levels of contamination in most sections are below the levels of mandatory clean-up of hazardous sediments. Sewer separation alone will not fully alleviate the problem of toxicity contained in urban runoff flows from separate stormwater drainage. Reducing toxicity in this prime spawning and fish propagation area located at the confluence of Hickory Creek and the Des Plaines River (Figure 1.3) is a key step for improving the biotic integrity of the entire Upper Dresden Island pool.

After improvements in the Hickory Creek, water quality and control of CSOs the prime habitat area should be remediated and, if necessary, toxic sediments in contaminated zones should be capped or the contaminated sediments should be removed.

Microbiological pollution - primary contact recreation. While the current general use standard for bacteria using fecal coliforms was not met, a low risk primary contact standard based on the new USEPA (2002) criteria is attainable. The Dresden Island Pool should not be considered as a prime zone for primary contact recreation, such recreational activities should be infrequent or accidental because of the effluent dominated nature of the river and the risks associated with navigation.

Remedial Action
To accomplish the goal of providing limited contact recreation in the Dresden Island pool, wastewater effluents discharging directly into the Dresden Island pool and Hickory Creek containing pathogenic microorganisms should be disinfected. The disinfection methods must be environmentally sensitive, such as chlorination followed by dechlorination or non-chlorine disinfection. Disinfection of effluents in the Chicago Area Waterways would not bring about a significant improvement in the Dresden Island pool due to die-off of bacteria during the time of travel. This issue as it pertains to the recreational use of the Chicago Area Waterways will be addressed in the subsequent UAA.

This action will bring the river into compliance with primary contact medium risk recreation standards that would allow and protect infrequent primary contact and also protect swimmers in the sections downstream of the I-55 bridge.

Temperature
Due to the heated discharges from the Joliet Power plant units, the temperature in the Dresden Island pool between the discharge of heated water and the I - 55 Bridge reaches levels that are lethal to fish. This was documented in the Burton's (1995) study that showed high mortality of fish (fathead minnow, *Pimethales promelas*) and benthic invertebrate (Scud-*Hyalella azteca*) at 35°C, which is less than the temperature measured in the stretch of the river between the thermal outfalls and the I-55 Bridge. Evidence provided by the Midwest Generations in the presentation to the biological expert subcommittee indicated that temperature in 1999 had exceeded the Secondary Contact and Indigenous Aquatic Life Standard. Also a compilation of temperatures lethal to fish (see Chapter 2) has shown that the lowest lethal temperatures for most common fish species are less than 37.8°C (100°F). Therefore, the Secondary Contact and Indigenous Aquatic Life Illinois standard does not protect the aquatic life in the stretch. Figures 2.44 and 2.45 also show that the General Use standard is protective of most adult fish population. Thus, implementing the General Use standard for temperature is a necessary step to improve the biotic integrity of the Upper Dresden Island pool to a level commensurate with the impounded water bodies with balanced biological communities.

It is also necessary to address the temperature differential between the intake of the river water to the power plants and the effluent during low flows.

Remedial Action
We believe that reduction of thermal loadings from the Joliet plants should be implemented that would bring the temperature in the Upper Dresden Island pool (between the heated discharges of the Joliet plants and the I - 55 Bridge) in compliance with the General Use standard. Whether this compliance with the General Use temperature standards will bring about a wide spread adverse socio economic impact on the utility and on the local area should be assessed in consultation with Midwest Generation and other stakeholders. While the General Use thermal standard is necessary and appropriate to protect the aquatic community otherwise attainable within the Upper Dresden Island pool, economic and operational considerations may be significant and should be given due consideration in the development of any alternate standards and the compliance period to attain that

new standard. The Agency should work closely with Midwest Generations and other affected thermal sources to accurately estimate the technical, financial and scheduling requirements. If attainment of the Illinois General Use Standard is found to cause a substantial and wide spread socio-economic impact, we recommend that a new standard include a maximum temperature that represents the upper bound to prevent lethality of known indigenous fish species and additional criteria to address general growth and health needs of aquatic life effects. Figures 2.44 and 2.45 clearly document that the current General Use thermal standards provide adequate protection to the potentially indigenous aquatic species that would reside in the Dresden Island pool and should, therefore, provide the reference level for the socio-economic study. This is also required by the Water Quality Standards regulations.

River Management Measures

We have pointed out that because of its heavy use for navigation and effluent domination, the river needs help and management. After the proposed remedial short term actions are implemented, we recommend the following management measures

- fish population management and restocking, considering the fact that the river will remain in the long-term enriched by nutrients
- providing fish passage between the pools
- control and prevention of sediment contamination
- turbine and dam aeration of the Brandon Road inflow
- provide warnings of water quality emergencies

Nutrient Enrichment Problem

An issue that was left behind and that could become a future water quality issue, because of the anoxia problem in the Gulf of Mexico and potential local problems, is a high level of nutrients in the Des Plaines River. We have addressed ammonium toxicity and found it not to be an issue of serious concern. However, nitrogen and phosphorus cause other problems that have not been adequately addressed by the regulatory agencies. These problems are:

A) Excessive algal development that may interfere with recreation and the aesthetic of the river.
B) Dissolved oxygen problem caused by photosynthesis and respiration that result in large daily fluctuations of the DO concentrations (see Chapter 2).

The US EPA and the Illinois EPA are working toward implementing workable nutrient standards. The issue of nutrient levels will be addressed when such standards become available. This UAA has found that, at this time, the elevated nutrient levels do not interfere in a major way with the attainment of the water quality goals for the Lower Des Plaines River as expressed by the Illinois General Use standards.

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