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A Screening Assessment of the Potential Impacts of Climate Change on the Costs of Implementing Water Quality-Based Effluent Limits at Publicly-Owned Treatment Works in the Great Lakes Region

External Review Draft Report

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PREFACE

The Environmental Protection Agency's Global Change Research Program (GCRP) is an assessment-oriented program within the Office of Research and Development that focuses on assessing how potential changes in climate and other global environmental stressors may impact water quality, air quality, aquatic ecosystems, and human health in the United States. The Program's focus on water quality is consistent with the *Research Strategy* of the U.S. Climate Change Research Program - the federal umbrella organization for climate change science in the U.S. government - and is responsive to EPA's mission and responsibilities as defined by the Clean Water Act and the Safe Drinking Water Act. The GCRP's water quality assessments also address an important research gap. In the 2001 *National Assessment of the Potential Consequences of Climate Change in the United States* (Gleick and Adams, 2000), water quality was addressed only in the context of the health risks associated with contaminated drinking water. A comprehensive assessment of the potential impacts of global change on water quality was not included.

Since 1998, the National Center for Environmental Assessment's office of the GCRP has assessed the consequences of global change on water quality. Through its assessment projects, this Program has provided timely scientific information to stakeholders and policy makers to support them as they decide whether and how to respond to the risks and opportunities presented by global change. This report assesses the potential effects of climate change on the costs implementing water quality-based effluent limits at publicly owned treatment works (wastewater treatment facilities) in the Great Lakes Region. Water treatment infrastructure was identified as a priority concern because water treatment is an essential service necessary to protect public health and ecosystems. Investments in water treatment infrastructure are also capital-intensive, long-term in nature, and irreversible in the short- to medium term. Decisions made today will thus influence the ability of treatment facilities to adapt to changes in climate for many years into the future.

The report is a screening level analysis intended to determine the scope and magnitude of global change impacts rather than a detailed assessment of specific impacts and adaptation measures. Together with a companion report addressing the potential effects of climate change on combined sewer overflow (CSO) events, this report fulfills a GCRP 2006 Annual Performance Measure to complete *"two external review draft reports detailing the possible impacts of global change on combined sewer overflows in key regions, and the possible effects of climate change and variability on operations and management of publicly operated treatment works (wastewater facilities) for OW and EPA Regions."*

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Executive Summary

Climate is a major factor influencing the amount, timing, and quality of water available meet human needs (Gleick 2000). During the last century, much of the U.S. experienced increased ambient air temperatures and altered precipitation patterns (NAST 2000). Projections of future climate suggest these trends are likely to continue, and potentially accelerate during the next century. Future changes in climate could thus impact water quality management.

Streamflow is strongly influenced by precipitation intensity and frequency, air temperature, and various natural and anthropogenic factors affecting watershed hydrologic processes. Projected impacts of climate change on streamflow include changes in both the total amount and temporal variability of flow. Climate change is expected to increase the proportion of rainfall occurring in high intensity events (US GCRP 2000), resulting in increased stormwater runoff and high flow events. At the same time, a shift towards more intense storms will decrease infiltration and groundwater recharge, resulting in reduced low flow periods between events.

Increased temperatures in the GLR will also result in increased evapotranspiration. As a result, more precipitation will be lost to the atmosphere and less will run off into streams and rivers, especially during the summer and early fall months when low flow periods are most likely to occur. Increased temperatures would also mean a shift in the form of precipitation – more would fall as rain and less as snow. As a result of this shift, and an earlier melt of the snowpack in the spring, streamflows during the summer are likely to be reduced in the GLR (US GCRP 2000).

Publicly owned treatment works (POTWs) discharge billions of gallons of effluent daily to receiving water bodies throughout the U.S. One of the principal pollutants associated with POTW effluent is organic matter. Naturally occurring microbial populations in receiving waters consume dissolved oxygen (DO) as they decompose organic matter. Low DO is a significant problem for aquatic ecosystems and is a leading cause of impairment for water bodies listed under Section 303(d) of the Clean Water Act (CWA).

The design characteristics of POTWs are directly tied to hydroclimatological metrics such as daily precipitation and receiving water low-flow conditions. Generally, these systems are designed to handle storm or flow events of a given intensity, duration, and frequency, and there is an implicit assumption that precipitation and flow are constant over time. Potential changes in climate could reduce low-flow conditions in receiving waters, reducing the dilution of effluent, and increasing the likelihood of DO impairment below POTW discharge locations.

Under the CWA, all point source discharges must obtain a National Pollutant Discharge Elimination System (NPDES) permit, which specifies limits on pollutant concentrations or loads in the discharge. If it is determined that a discharge has the reasonable potential to cause or contribute to an exceedance of a water quality standard (WQS), the discharger's NPDES permit must contain a water quality based effluent limit (WQBEL) for that pollutant. Current guidance established by the EPA's Office of Water for implementing WQBELs recommends the use of the 7-day averaging period, 10-year recurrence interval low flow (7Q10) as the design flow for establishing assimilative capacity and WLAs for a water body. If climate change results in reductions in low flow volume, and therefore assimilative capacity, WQBELs would need to be more stringent, and treatment costs would increase.

The Intergovernmental Panel on Climate Change (IPCC) Regional Assessment of Vulnerability for North America (Mulholland and Sale 1998) found that climate change may lead to a reduction in summer baseflows, which will reduce the assimilative capacity for wastewater effluents and exacerbate existing or produce new water quality problems. In the Great Lakes Region, it has also been estimated that climate change could decrease annual freshwater flows into the Great Lakes from streams and rivers by 20% (US EPA 2001). It is thus possible that low flow conditions, including 7Q10 events, will decrease throughout the region.

The objective of this research was to characterize the scope and magnitude of climate change impacts on operating costs[1] at POTWs discharging to rivers and streams in the Great Lakes Region (GLR). This effort was designed to be a screening study, focusing on costs of treating a single pollutant (BOD_5), from a single point source category (POTWs), for only those facilities in the GLR most likely to be subject to water quality based effluent limits (POTWs discharging to water bodies listed as impaired for DO or related impairments). The study focused on 147 POTWs identified as discharging to impaired receiving waters in the GLR.

Although WQBELs have been established for many dischargers, it is still the case that even where WQBELs are appropriate, many permits are still based on technology-based effluent limits (TBELs). Thus, before estimating the cost of more stringent treatment standards associated with climate change, it was first necessary to estimate the treatment efficiency (and associated cost) required for full implementation of WQBELs under current climate conditions. Then, starting from the baseline conditions, we estimated the cost of more stringent WQBELs resulting from potential climate change-related decreases in receiving stream flow.

A bounding analysis framework was used to evaluate scenarios for (1) treatment requirements associated with WQBEL implementation under current climatic conditions (referred to as BL-WQBEL) and (2) more stringent treatment requirements associated with climate change (referred to as CC-WQBEL). Two BL-WQBEL scenarios, (assuming more and less stringent requirements to meet WQBELs), and two CC-WQBEL scenarios (assuming high and low estimates of decreases in 7Q10) were assessed.

Based on two case studies of TMDL implementation, values of 12 percent and 50 percent were chosen to represent the range of incremental increase in BOD_5 removal efficiency, beyond TBELs, that will typically need to be attained by POTWs to meet WQBELs in reaches impaired for dissolved oxygen (*i.e.*, the BL-WQBEL). Based on an EPA estimate of potential reductions in annual streamflow in the GLR (US EPA 2001), and a study by Eheart et al. (1999) of potential climate change-related effects on the Sangamon River (IL), values of 20 percent and 57 percent were chosen to represent low and high decrements in 7Q10 low flow events resulting from climate change. The incremental changes in WQBELs required due to climate change (CC-WQBEL) were assumed to be linearly proportional to changes in streamflow.

A cost model was developed to translate the estimates of BOD_5 removal efficiency into estimates of annual cost. The cost model provided an estimate of cost versus removal efficiency as a function of POTW size (effluent flow). The annual cost of achieving the high and low range values for BL-WQBEL and CC-WQBEL was then estimated for each of the 147 POTWs, and expressed in terms of incremental cost.

[1] Throughout this report, the term “operating costs" is used to represent annualized capital plus operating and maintenance (O&M) costs.

Results indicate that if WQBELs were implemented under current climate conditions, costs summed across all 147 POTWs are estimated to increase by $14 million (= 4%) to $59 million (= 17%) per year over TBEL levels (for the low end and high end, respectively). This is equivalent to an average annual cost increase of $95,000 to $400,000 per facility over TBEL levels. Accounting for climate change (i.e. due to reduced 7Q10 streamflow in receiving waters) would increase the incremental cost of implementing WQBELs summed across all 147 POTWs by an additional $8 million (= 2%) to $97 million (= 24%) per year over the current cost of implementing WQBELs. This is equivalent to an average annual cost increase of $54,000 to $660,000 per facility over the current cost of implementing WQBELs.

As a general screening analysis, these results suggest that climate change could have a significant effect on two of EPA's most important water programs – NPDES permitting and POTW financing through the State Revolving Fund (SRF). Given the limited scope of the analysis (e.g. it addresses only one pollutant, one source type, one region, and only includes POTWs discharging to impaired streams) – it is likely that these cost estimates do not reflect the full scope of climate change impacts on WQBEL implementation.

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1. Introduction

Climate is a major factor influencing the amount, timing, and quality of water available meet human needs (Gleick 2000). During the last century, much of the U.S. experienced increased ambient air temperatures, sea level rise, and altered precipitation patterns (NAST 2000). Projections of future climate suggest these changes are likely to continue, and potentially accelerate during the next century. Future changes in climate thus could have a major impact on water quality management.

Water quality management infrastructure typically have long lifetimes, significant capital costs, and design characteristics that are directly tied to hydroclimatological metrics such as daily precipitation and receiving water low-flow conditions. Generally, these systems are designed to handle storm or flow events of a given intensity, duration, and frequency, and there is an implicit assumption that precipitation and flow are constant over time.

Publicly owned treatment works (POTWs) are examples of such water quality management systems. POTWs have a significant impact on water quality because they discharge billions of gallons of effluent daily to receiving water bodies throughout the U.S. Potential changes in climate could reduce low-flow conditions in receiving waters, reducing the dilution of effluent, and increasing the likelihood of water quality impairment below POTW discharge locations.

Under the CWA, all point source discharges must obtain a National Pollutant Discharge Elimination System (NPDES) permit, which specifies limits on pollutant concentrations or loads in the discharge. If it is determined that a discharge has the reasonable potential to cause or contribute to an exceedance of a water quality standard (WQS), the discharger's NPDES permit must contain a water quality based effluent limit (WQBEL) for that pollutant. Section 303 also establishes the Total Maximum Daily Load (TMDL) program, which requires states to determine the maximum allowable amount of a pollutant an impaired water body can receive and still meet water quality standards. WQBELs may be established regardless of whether a TMDL has been developed for the receiving stream, but if a TMDL has been developed, WQBELs are the regulatory mechanism by which waste load allocations (WLAs) for point source dischargers are implemented.

WQBELs are developed under the assumption that pollutant loads should not exceed the assimilative capacity of streams during low-flow conditions. Current guidance established by the EPA's Office of Water for implementing WQBELs and TMDLs recommends the use of the 7-day averaging period, 10-year recurrence interval low flow (7Q10) as the design flow for establishing assimilative capacity and WLAs for a water body. Future increases in temperature together with changes in the pattern of rainfall are likely to result in decreased low flow events in much of the U.S. If climate change results in reductions in low flow volume, and therefore assimilative capacity, WQBELs would need to be more stringent, and treatment costs would increase.

The objective of this research was to characterize the scope and magnitude of climate change impacts on operating costs[2] at POTWs in the Great Lakes Region (GLR). This effort was

[2] Throughout this report, the term "operating costs" is used to represent annualized capital plus operating and maintenance (O&M) costs.

designed to be a screening study, focusing on costs of treating a single pollutant (BOD_5), from a single point source category (POTWs), for only those facilities in the GLR most likely to be subject to water quality based effluent limits (POTWs discharging to water bodies listed as impaired for DO or related impairments). The study also addressed only facilities discharging to rivers and streams whose discharge limits are sensitive to changes in low flow conditions, (facilities discharging to lakes were not included).

This line of investigation is timely given the gap between anticipated funds available and funds needed to finance improvements in wastewater treatment infrastructure. The Clean Water Needs Survey (CWNS) released in 2004 indicates that $57 billion will be required for improving wastewater treatment systems in the U.S. over the next 20 years (US EPA 2003), well beyond the funds available through the State Revolving Fund (SRF), the principal source of funds for these improvements. The extent to which climate change could widen this funding gap is an important issue for EPA as well as state and local governments.

1.1 WQBELs and the TMDL Program

In this section, we describe some of the key features of WQBELs, focusing on their relationship with the TMDL program. In many cases, WQBELs are in place because a TMDL has been developed. WQBELs may be required, however, regardless of whether a TMDL has been developed for the receiving stream (or whether the receiving water body has been listed as impaired). Assumptions about WQBEL implementation in this study are related to the TMDL program in the following way:

- Estimates of the baseline treatment efficiency to meet WQBELs are based on data from the TMDL program
- The set of POTWs analyzed were selected because they discharge to impaired receiving waters where TMDLs have been or will be developed

1.1.1 Watersheds and Management Practices

The formulation of water quality management plans in the United States is driven by U.S. EPA rules and guidance for the determination of TMDLs, a calculation of the maximum amount of a pollutant that a water body can receive and still meet water quality standards. The Clean Water Act, section 303, requires that states establish water quality standards, and if standards are not met, then a TMDL must be developed.

The TMDL process integrates and evaluates all potential sources of a pollutant impacting a water body, and is a water quality-based approach to implementing water quality standards. It is applied to an entire watershed or drainage basin whenever possible, but may also be applied to water body segments with individual or multiple pollutant sources. The TMDL process provides the basis for determining whether a proposed discharge of a pollutant has the potential to cause or contribute to an excursion of water quality standards. If a discharge poses a reasonable potential for exceeding a standard, the TMDL process is used to determine the WQBELs for all sources of that pollutant to assure compliance with water quality standards. If not, the watershed or water body segment is not water quality limited for that pollutant (Bromberg 1996).

Under the CWA, all point source discharges must obtain a National Pollutant Discharge Elimination System (NPDES) permit, which specifies limits on pollutant concentrations or loads in the discharge. For those waters not listed as impaired, WQBELs may still be imposed through the NPDES permitting process. If technology-based effluent limits (TBELS) are deemed insufficient to attain water quality goals, the CWA (section 303(b)(1)(c)) and NPDES regulations (40 CFR 122.44 (d)) require that the permit writers develop more stringent WQBELs designed to ensure that water quality standards are attained (US EPA 1984).

1.1.2 Designation of Water Quality Limited Stream Reaches (303(d)) list

Section 303(d) of the Clean Water Act (CWA) requires that states, territories, and authorized tribes create lists of impaired waters. For these impaired waters, each state must establish a TMDL which specifies the point and non-point source pollutant loadings which will bring the water body into compliance. Currently, these entities must submit to EPA a complete listing of impaired waters within their jurisdictions every two years. A “303(d) impaired water body” is one that is not meeting Water Quality Standards (WQS) despite implementation of existing pollution control measures.

The 303(d) listings and methodology must be approved by the EPA. If EPA determines that the states, territories, and authorized tribes are not making progress towards the creation of a 303(d) list, they can intervene and complete the listing process. After the impaired listing process, these jurisdictions generally have 8-13 years to establish a complete set of TMDLs for a given reach (US EPA 2002b).

1.1.3 Approaches to Allocate Loads

The WQS for a water body are based on its designated use classification. A TMDL is the maximum amount of a given pollutant that a water body can receive and still meet the WQS for its designated use. The TMDL is the sum of loadings from point sources (such as POTWs and industrial facilities) and non-point sources (such as urban and agricultural runoff, and natural sources). TMDLs are required to contain a margin of safety to account for modeling and measurement error and seasonal variations, which in most cases can be integrated into the TMDL by using conservative estimates for the waste load allocation (WLA) for point sources and load allocation (LA) for non-point sources. TMDLs are calculated according to the following formula:

$$\mathbf{TMDL = \Sigma WLA + \Sigma LA + MOS}$$

Where:

WLA = Wasteload allocation from point sources

LA = Load allocation from non-point sources, includes natural loadings

MOS = Margin of safety

Allocating the load reductions across sources is considered by many to be the most challenging aspect of the TMDL program, with many complex technical and equity issues. There are 19 different allocation schemes that can be used, ranging from equal load allocation to equal cost allocation (US EPA 1991). In the NPDES permitting process, the WLA for a POTW becomes the basis for the legally enforceable WQBEL. The following are some of the factors that make load allocation technically challenging:

- The objective is generally to meet a water quality standard at all points throughout the stream reach. Water quality is rarely uniform throughout the reach, however, and some sources usually have a greater influence than others on the locations with poorest water quality.

- A unit of load reduction for one source may not be technically or economically feasible at another source.

One of the key factors driving a WQBEL determination for a stream or river is the amount of water available to assimilate the pollutant load. For most pollutants, critical loads are determined for a low-flow design period, during which the in-stream dilution provided by the waterway is at a minimum. EPA recommends a design upstream flow for acute aquatic life criteria at the 1Q10 (1-day low flow over a 10-year period) and for chronic aquatic life criteria at the 7Q10 (7-day low flow over a 10-year period) (US EPA 1995; US EPA 2006).

Dissolved oxygen (DO) is a key indicator of the health of a water body which, when depleted, can pose chronic health effects to aquatic ecosystems. Concentrations of DO in the water are reduced by the presence of organic matter, which is consumed by naturally occurring aerobic microbes in the water. Nutrients in the water, such as nitrogen and phosphorus, also influence DO concentrations through the facilitation of algal growth, which consumes oxygen at night during the respiration phase of the photosynthesis-respiration cycle. A typical DO water quality standard to sustain fish and aquatic life is 5 mg/l, though this may be set higher or lower depending on state water quality goals and the designated use of the water body (US EPA 2000). Because reduced DO results in chronic effects, the low-flow design period used for this study is the 7Q10 low-flow event.

1.2 Municipal Wastewater Treatment

1.2.1 Municipal Wastewater Pollutants

The primary contaminant in municipal wastewater is organic matter. When organic matter reaches a stream or river, the decomposition process reduces the aqueous concentration of DO. Increases in air temperature may induce a shift in aquatic biota by increasing surface water temperatures, resulting in lower DO saturation concentrations and increased rates of BOD decomposition (Morill et al. 2005). The standard measurement for the level of organic matter in municipal wastewater is the 5-day biochemical oxygen demand test (BOD_5). This is a measure of the dissolved oxygen consumption for a known volume of wastewater over a 5-day incubation period at 20^{o} C.

In 303(d) reaches impaired for low DO, TMDLs are typically specified in terms of BOD_5 loadings. Although in some cases excessive nitrogen or phosphorous loadings may be the cause of DO impairment, in this study the focus is only on BOD_5. The average influent BOD_5 concentration for municipal wastewater is 215 mg/l (US EPA 1998b), and the wastewater can be treated through several processes.

There are many other pollutants of concern in POTW effluent, most commonly suspended solids and microbial pathogens; however, the scope of this study is limited to BOD_5.

1.2.2 Overview of Typical Treatment Systems

The BOD_5 treatment process at POTWs is organized into a stepwise regimen of primary, secondary, and advanced levels.[3] Primary treatment generally consists of a screening process to remove large objects from the influent and is then followed by a settling tank, which allows suspended solids to settle out and be removed as sludge. This level of treatment has an average BOD_5 removal efficiency of approximately 43%, which varies depending on influent and plant characteristics (US EPA 1998b).

Secondary treatment is typically comprised of an activated sludge system that follows primary treatment. After primary treatment the influent goes through an aeration process to facilitate aerobic bacterial digestion, and then a secondary settling tank for the further removal of suspended solids. Often, a portion of these solids is returned to the influent at the aeration process to further facilitate the bacterial digestion process (the remaining solids are managed as biosolids or sludge). Combined with primary treatment, at the end of this step the average BOD_5 removal efficiency is about 85%, but again the level varies depending on influent and plant characteristics (US EPA 1998b).

There are also advanced primary and advanced secondary treatment systems where various chemical steps can be added to increase BOD_5 removal efficiency of these processes, generally by increasing settling rates. Advanced primary treatment can reach removal efficiencies close to secondary, and advanced secondary treatment can exceed 90% removal efficiency. Advanced treatment to further reduce BOD_5 concentrations beyond that of advanced secondary can include filtration and reverse osmosis, and can reach BOD_5 removal efficiencies of 95% and beyond (Metcalf and Eddy 2003).

1.2.3 Effluent Guidelines

Great strides have been made over the last 30 years in the average level of municipal wastewater treatment as a result of the combination of federal regulations for POTWs to meet secondary treatment standards (see 40 CFR Part 133) and the more than $60 billion in grants from the federal government for treatment plant construction and other related projects (US EPA 1998c). Under the secondary treatment standards, discharge permits must require POTWs to meet a BOD effluent standard of 30 mg/l and a BOD5 removal efficiency standard of 85%. There are some allowances for reduced levels of treatment that are granted on a case-by-case basis (*e.g.*, situations where pre-existing treatment is close to these standards, episodic high flow during rain events, low BOD_5 influent concentration, or extreme geographic and climatic characteristics). In such situations, allowances can only be made if it is determined that the increased pollution will not adversely impact the receiving body of water (US EPA 1984).

As a result of some POTWs operating below the CWA standards, and some operating above them, the average level of treatment for all POTWs throughout the nation is 85% removal efficiency of BOD_5 (US EPA 1998b). Compared to historical levels, the reduced BOD_5 loading to our nation's streams and rivers has produced an increase in DO levels, and thus an

[3] It should be noted that there are other treatment processes, such as chlorination (to treat bacteria) and sludge processing, which are not discussed in this memo because they do not directly relate to BOD_5 removal and effluent quality.

improvement in overall water quality. Of a set of 311 impaired reaches below POTW outfalls monitored by the EPA throughout the US, 69% showed an improvement in worst-case DO levels, with an increase in minimum DO levels from an average of 4.1 to 7.2 mg/l for the top 25 improving reaches (US EPA 1998b). However, even with this progress, there are many streams and rivers that are still impaired. The current implementation of TMDLs for bodies of water throughout the nation is a comprehensive attempt to improve water quality in thousands of stream reaches, and it will require treatment beyond secondary treatment standards for thousands of POTWs.

1.3 Impacts of Climate Change on the Hydrologic Cycle

Streamflow is strongly influenced by precipitation intensity and frequency, air temperature, and various natural and anthropogenic factors affecting watershed hydrologic processes. Projected changes in climate are thus likely to have a significant impact on the amount and seasonal variability of streamflow.

Climate change is expected to increase the proportion of rainfall occurring in high intensity events (US GCRP 2000). This will tend to increase stormwater runoff during events, resulting in increased high flow events without necessarily increasing the amount of water available. A shift towards more intense storms will also decrease infiltration and groundwater recharge. Low flow periods occur when there is little or no precipitation, and streamflow is supplied by groundwater. If a higher proportion of total precipitation is delivered in intense events, unless total precipitation increases significantly, low flow periods between events will thus likely decline.

Increased temperatures in the GLR will result in increased evapotranspiration. As a result, more precipitation will be lost to the atmosphere and less will run off into streams and rivers, especially during the summer and early fall months when low flow periods are most likely to occur. Increased temperatures would also mean a shift in the form of precipitation – more would fall as rain and less as snow. As a result of this shift, and an earlier melt of the snowpack in the spring, streamflows during the summer are likely to be reduced in the GLR (US GCRP 2000).

In many areas of the US, groundwater levels are predicted to decline over the next 100 years as a result of climate change (US GCRP 2000). A reduction in groundwater levels would have the greatest impact during the dry summer season when many streams and rivers are dependent on groundwater as a source of baseflow.

Extreme low flow conditions tend to be correlated with poor water quality due to reduced dilution of point source pollutants in receiving waters (US EPA 1998a). The Intergovernmental Panel on Climate Change (IPCC) Regional Assessment of Vulnerability for North America (Mulholland and Sale 1998) found that climate change may lead to a reduction in summer baseflows, which will reduce the assimilative capacity for wastewater effluents and exacerbate existing or produce new water quality problems. In general, water quality problems associated with human impacts on water resources via wastewater discharges (*e.g.*, low dissolved oxygen levels and high contaminant concentrations resulting from wastewater effluents) will be exacerbated more by reductions in streamflow, particularly during summer baseflow periods, than by other changes in hydrologic regimes (Mulholland and Sale 1998). More severe droughts, particularly in summer, could result in reduced water quality (*e.g.*, lower dissolved oxygen [DO] concentrations, reduced dilution of effluents) and impaired habitat (*e.g.*, drying of streams, expansion of zones with low dissolved oxygen concentrations, water temperatures exceeding thermal tolerances).

1.4 Climate Change Adaptation and Water Resource Decision-making

To focus decision support resources where they will have the most benefit, it is helpful to identify situations where decisions are sensitive to climate-related factors, and where significant resources will be invested (*e.g.,* Purkey et al. in press; Freed and Sussman in press).

Based on these criteria, the TMDL program is a good candidate for decision support because:

1. Decisions are often dependent on climate-sensitive flow parameters, such as the frequency or severity of low-flow conditions;
2. The establishment and implementation of TMDLs is a high-priority for state and federal partners; and
3. TMDLs are associated with long-term, investments that are expected to perform for decades into the future.

Furthermore, the focus of this study on one category of discharger, POTWs, is warranted because:

1. POTWs are a major source of point source discharges throughout the U.S.;
2. POTWs are managed by public agencies through decision-making processes that can be influenced by EPA;
3. Many POTWs have relatively similar treatment processes (and thus can be analyzed in a relatively straightforward way); and
4. POTWs are the subject of strategic analysis at the Agency due to the large gap in SRF available versus funds needed for treatment improvements.

Although relevant to many parts of the United States, this study focuses on POTWs in the Great Lakes Region (GLR). The focus on the GLR results from EPA's long-term interest in this area through its responsibilities to the first national assessment of the impacts of climate change on the United States (US GCRP, 2000), and because EPA maintains strong working relationships with key stakeholders in the region.

2. Methods

The methodology for estimating the potential impacts of climate change on the cost of implementing WQBELs started with identifying POTWs on impaired reaches using the 303(d) listings for each GLR state and the industrial facilities database (IFD). This process identified a set of 147 POTWs to be included in the analysis. The next step was to estimate treatment costs for these POTWs at varying levels of BOD_5 removal efficiency. A cost-treatment efficiency curve for BOD_5 removal that could be scaled to various sizes of POTWs was developed using published POTW operation cost functions for capital and operations and maintenance expenditures (NRC 1993, Qasim 1999). By combining these cost functions, a model was created to estimate operating costs for a range of POTW capacities that was inclusive of all active POTWs identified using the IFD and 303(d) listings.

Although WQBELs have been established for many dischargers, it is still the case that even where WQBELs are appropriate, many permits are still based on technology-based effluent limits (TBELs). Thus, before estimating the incremental cost of more stringent treatment standards associated with climate change, we had to first estimate the treatment efficiency (and associated cost) required for full implementation of WQBELs under current climate conditions. Then, starting from the baseline conditions, we estimated the cost of more stringent WQBELs resulting from potential climate change-related decreases in receiving stream flow.

Establishing WQBELs is a complex and site-specific activity. To simplify the system and make the analysis tractable, we applied a bounding analysis framework to evaluate combinations of scenarios for (1) treatment requirements associated with baseline WQBEL implementation (referred to as BL-WQBEL) and (2) more stringent treatment requirements associated with climate change (referred to as CC-WQBEL). We developed two BL-WQBEL scenarios (less stringent/ more stringent) and two CC-WQBEL scenarios (low decrement in 7Q10/ high decrement in 7Q10), for a total of four combinations, and analyzed costs for each combination.

2.1 POTW Identification

2.1.1 Water Quality Limited Stream Reaches (303(d) list)

Initially, all of the POTWs in the GLR that discharge into receiving waters with existing water quality impairment were considered for inclusion this study. This initial list was pared down to a smaller set using criteria described below. The Great Lakes Region was defined using the border established by the Great Lakes Regional Climate Change Assessment (see http://www.geo.msu.edu/glra/region/region.html). This region includes the states of Michigan, Minnesota and Wisconsin in their entirety, as well as counties within Illinois, Indiana, New York, Ohio, and Pennsylvania that border the Great Lakes.

Impaired reaches were defined as those listed on the 303(d) list of impaired waters developed by each state as required under the CWA. The 303(d) database includes data on the basic attributes for each impaired reach, as well as the location of the reach (i.e., the spatial element). Multiple reach records exist where more than one impairment has been identified for a reach segment. The impairment type was linked to the spatial element by linking the LIST_ID field of the impairment list, which includes the impairment type(s) according to EPA and the state for each reach segment, to the ENTITY_ID of the 303(d) table.

The focus of this assessment was narrowed to those impairment types related to dissolved oxygen (DO) levels. The relevant impairment type as listed by EPA is "Organic Enrichment/Low DO." Corresponding state impairment types are:

-Biochemical Oxygen Demand	-Low Oxygen
-Biological Oxygen Demand	-Organic Enrichment
-BOD	-Organic Enrichment / Low Dissolved Oxygen
-BOD_5	-Organic Enrichment / Low DO
-CBOD	-Oxygen Demand
-Chemical Oxygen Demand	-Sediment Oxygen Demand
-Dissolved Oxygen	-Sludge / Sediment
-Eutrophic	-TOC Trend
-Eutrophication	-TSI
-Hypoxia	-Low Dissolved Oxygen

The list of water quality-limited reaches was further narrowed to include only creeks, streams, and rivers, as the calculation of assimilative capacity in lakes is not related to changes in low-flow conditions. It should be noted that some of the larger POTWs in the GLR discharge directly to the Great Lakes. Given that POTWs discharging to lakes were omitted from this study, it is likely that there was an undersampling of the largest POTWs in the region.

This screening resulted in the identification of 2,351 reach segments water quality-limited for DO within the study area. However, many of these records represent segments of the same river or stream reach. When grouped by Reach ID (from the "ReachFile 1" data set), the number of reach segments was reduced to fewer than 250 individual existing impaired reaches.

2.1.2 POTWs on DO-impaired Reaches

Given the list of 250 DO-impaired reaches in the GLR, the Industrial Facilities Discharge (IFD) database was queried to pinpoint POTWs[4] discharging to these waters. This database contains spatial locations for industrial or municipal point sources discharging to surface waters. These data include multiple SIC (Standard Industrial Classification) codes for each facility, of which the code "4952" was used to identify POTWs.

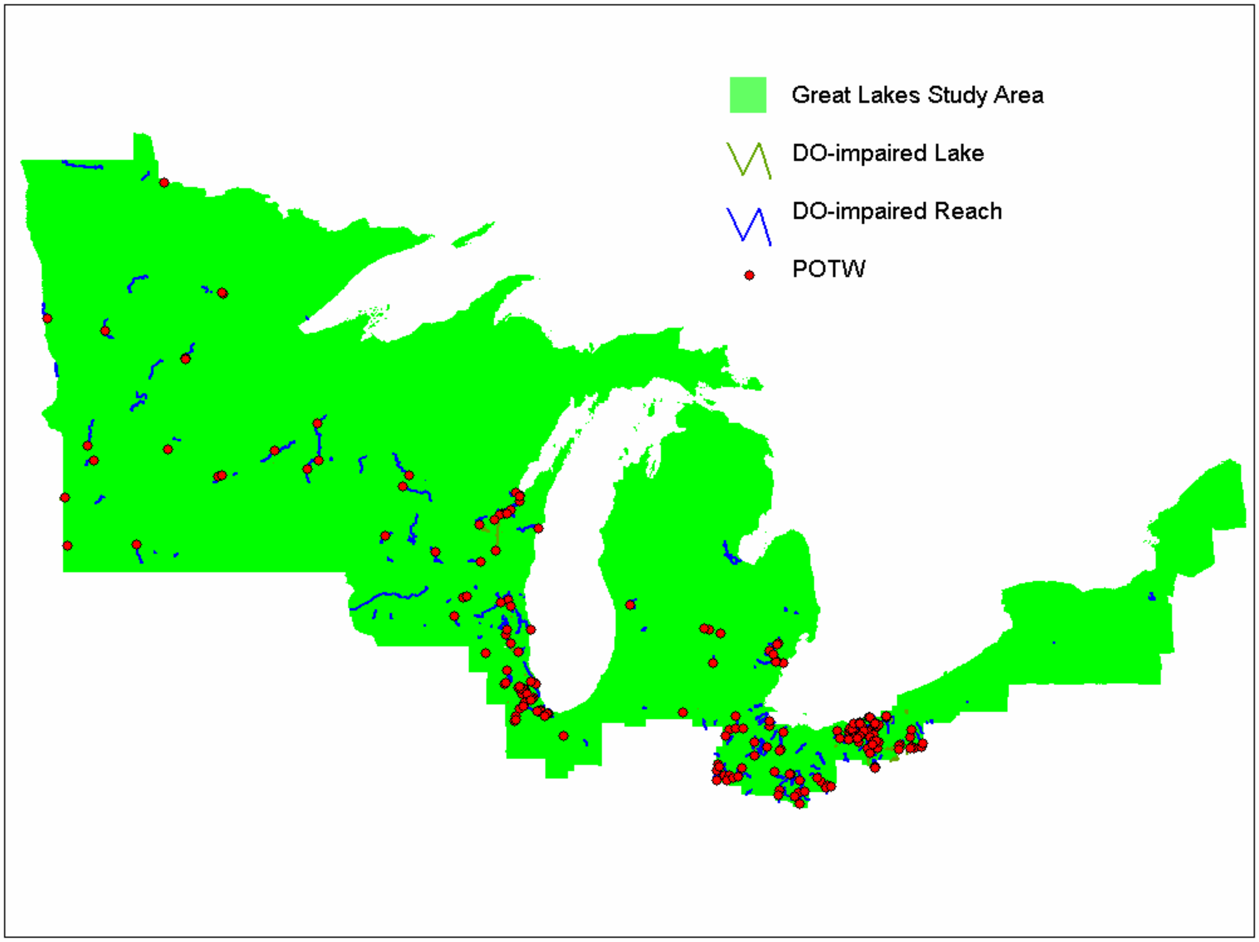
Facilities were identified with primary and secondary SIC codes equal to "4952." A half mile buffer was then created around DO-impaired reaches using a GIS (ESRI ARCGIS), and all POTWs located within this half-mile buffer were identified. This resulted in the selection of 235 POTWs located within a half-mile of a DO-impaired stream reach. Nine POTWs discharging to impaired lakes were eliminated from the analysis.

Due to error associated with the spatial locations of the facility database, this number provides a reasonable approximation, but should not be considered precise. Additionally, the buffer does not take into account the relative position of the facility to the stream, viz., whether it is downstream or upstream or whether it may be releasing directly to the stream or to a tributary of the stream.

[4] This database can be found on the EPA Office of Water BASINS CDROM series with documentation available online at http://www.epa.gov/waterscience/basins/metadata/ifd.htm.

Figure 1 shows the location of POTWs within a half mile of an impaired reach within the GLR. Of this set, only 184 POTWs were determined to be active. Thirty seven of the 184 active sites were very small (discharge less than 0.1 MGD), and were excluded from analysis due to concerns that the cost model would not be valid for very low discharge rates. This resulted in a set of 147 POTWs that were selected for inclusion in the study.

Figure 1. POTWs on Stream and River Reaches Impaired by Low DO in the Great Lakes Region



2.2 Development of the Cost Curve

To estimate the cost associated with incremental POTW treatment efficiency improvements, a cost curve was developed expressing the unit cost of treatment (including annualized capital costs and operating and maintenance (O+M) costs) per 1000 gallons treated for operating a POTW at a given level of treatment (expressed as percent removal of BOD_5). Capital costs include the annual costs of loan repayment for construction and equipment needs for the POTW. Operation and maintenance (O+M) costs consist of employee salaries, general upkeep of the plant, sludge disposal, electricity consumption, and other expenditures that occur on a regular basis that are essential for plant operation.

2.2.1 National Research Council Equations

The cost curve was estimated using data published in 1993 by the National Research Council (NRC), based on an extensive POTW survey conducted by MIT in 1990, and another survey conducted by the NRC in 1991 (NRC 1993). These surveys were combined with expert technical analysis to produce estimated annual capital and O+M costs for multiple treatment scenarios and their respective BOD_5 removal efficiencies (NRC 1993). The assumptions used by the NRC in determining these costs were an interest rate of 8% for a 20 million gallon per day (MGD) facility and a design lifetime of 20 years. Land costs were excluded from the capital cost functions. For this study, cost estimates were adjusted to 2006 dollars to account for inflation.

The NRC identified 10 types of treatment systems, and estimated the cost for each, as well as the removal efficiency for BOD, suspended solids, phosphorus, nitrogen, and other pollutants. The systems range from primary treatment only (system 1) to a system with chemically enhanced primary treatment, biological treatment, nutrient removal, high-dose lime treatment, filtration, granular activated carbon, and reverse osmosis (system 10). These systems span BOD_5 removal efficiencies of 30 to 100 percent. Significant performance data exist only for Systems 1 through 4, and are well characterized in the two POTW surveys. Systems 5 through 10 are uncommon, and the treatment efficiency and costs for those systems were based on literature reviews and professional judgment (NRC 1993). Several of these more advanced (and costly) systems are designed primarily to remove pollutants other than BOD_5, however, and thus are not viable options from a cost-effectiveness standpoint for reducing BOD_5. Thus only 4 systems were considered as alternatives for managing BOD in this study.

The systems corresponding to the range of interest for BOD_5 removal efficiency are listed in Table 1, and have removal efficiencies ranging from 78% to 98% removal. Table 1 also lists the annual costs for the treatment systems; the NRC provided the costs as a range, and the costs shown represent the midpoint of the range. The cost model uses an important assumption: *within each interval of treatment efficiency, it was assumed that the cost of BOD_5 removal is linear between each point generating a continuous function, i.e., there is a continuous rate of cost increase as removal efficiency increases (rather than a stepwise function[5])*. The cost curve is shown in Figure 2, which also includes points for two treatment systems outside the 78% to 98% range of interest in this study.

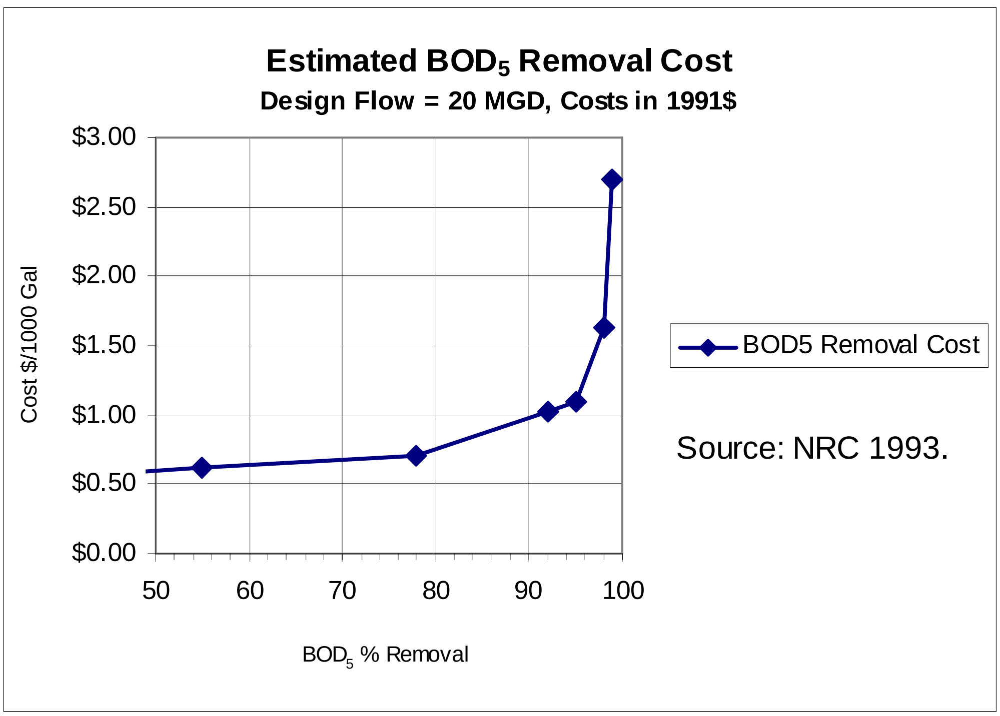
[5] The results of the analysis could be quite different if the removal efficiency increases were handled as step functions, though it is hard to estimate the net effect of such a change in methodology.

Table 1. Wastewater Treatment Cost and Level of Treatment for a 20 MGD POTW

Treatment System (and NRC System Number)	BOD_5 Removal Efficiency	Total Annual Cost (1991$ per Million Gal treated)
High-dose Chemical Primary (2b)	78%	$700
Conventional Primary plus Biological (3)	92%	$1,030
Chemically-enhanced Primary plus Biological Treatment (4)	95%	$1,100
Nutrient Removal plus Gravity Filtration (6)	98%	$1,625

Source: (NRC 1993).

Figure 2. Cost Curve for a 20 MGD POTW



2.2.2 Scaling Equations

The NRC cost functions supply useful information on the cost of various treatments for a given design flow (20 MGD), but the NRC report does not provide information that can be used to estimate costs for different plant capacities. Due to the influence of economies of scale, the treatment costs for smaller plants are higher than those for larger plants on the basis of dollars per 1000 gallons treated. To estimate costs for a range of plant sizes, scaling factors were derived based on a series of 28 cost approximation equations for POTWs (Qasim 1999). These equations were largely derived from an EPA report on municipal wastewater treatment (US EPA, 1976). For each of the cost elements, which range from activated sludge treatment to support personnel, a pair of equations is presented – one for capital cost and one for annual O+M cost. All of the cost elements are a function of daily design flow (Q); in many cases, the functions are exponential, with the exponents ranging from 0.43 to 2.

Scaling factors using the Qasim equations were developed for each of the four NRC treatment systems, across the range of plant capacities for the set of 147 POTWs in our study (from 0.1 MGD to 10 MGD). There were an additional 37 active POTWs with flows less than 0.1 MGD that were not evaluated because they were outside the functional range of the Qasim equations. Scaling factors were determined for each plant as follows:

1. The capital costs for construction were annualized using the same approach as NRC, *i.e.*, assuming an interest rate of 8% and a lifetime of 20 years.
2. The total annual costs were calculated as the sum of annualized capital plus annual O+M cost, for each cost element, for 20 different design flows (19 within the range of POTWs within the sample (viz., 0.1 to 10 MGD) and 20 MGD, the value provided by NRC).
3. From the 28 Qasim cost elements, those that were likely to correspond to each of the four NRC treatment systems were selected.
4. For each of the four treatment systems that mirrored the NRC systems, the costs were summed across the relevant cost elements for each of the 20 design flows. For example, the conventional primary plus biological system has 13 cost elements; summing the total system cost for plants with capacity of 1 MGD and 20 MGD, the results are $732 per year per 1000 gallons design flow and $209 per year per 1000 gallons design flow, respectively.
5. A "scaling factor" was developed throughout the flow range of interest, equal to the ratio of cost at a given design flow to cost for a 20 MGD plant. For the example above, the scaling factor is 3.51 (=$732/$209). The scaling factor was then multiplied by the NRC cost estimate for the corresponding treatment system. In this example, the NRC cost estimate for a conventional primary plus biological 20 MGD plant, adjusted to 2006$, is $1.54 per 1000 gal treated. The scaled unit cost for a 1 MGD plant would be 3.51 * $1.36 = $5.40 per 1000 gal treated.
6. For each of the 147 POTWs, the treatment cost was calculated by continuous interpolation across the range of treatment efficiencies, and multiplying flow by the unit cost. For example, to estimate treatment cost for a 1.2 MGD plant for a target treatment efficiency of 85%, the first step is to identify the "reference" flow (1 MGD is the closest flow category calculated). The two treatment systems whose efficiency brackets the target were then identified. In this case it would be high-dose chemical primary, at 78% (with a cost of $3.72 per 1000 gal treated), and primary plus biological, at 92% (with a cost of $5.39 per 1000 gal treated). The unit cost for 85% was determined by interpolating ($4.35 per 1000 gal treated).
7. The annual cost for each POTW was estimated by multiplying unit cost by flow (in thousand gallons). In the above example with flow of 1.2 MGD, annual cost would be $1.91 million (= 1,200 thousand gal/day * $4.35 per thousand gal treated * 365 days/yr).

The actual costs of increasing removal efficiency will vary from plant to plant based on existing POTW infrastructure and many complex, site-specific factors.

2.2.3 Characterizing POTW Rates of Discharge

Of the 184 active POTWs in the study set, 166 had average daily effluent discharge values reported in the IFD. For each of these 166 POTWs, the cost estimates were based on the reported discharge rates.

For the remaining 18 active POTWs with no data on discharge rates, flow was estimated as follows. Most of the active POTWs in the IFD database (including all 18 with no flow data) are listed as either major or minor. It is generally the case that major POTWs have average daily discharge rates greater than 1 MGD, while minor plants have average daily flows less than 1 MGD (US EPA, 1998c). There were 1 major and 17 minor plants for which there were no flow values reported. The mean flow was calculated for each of the two sets (major and minor) with flow data, and we assumed that the unknown plants have the mean flow for their set. For reference, the total flow across all of the POTWs in the study set was considerable. Including the 18 for which no flow data were available, total flow for the 184 active POTWs in the GLR, in water bodies impaired due to "Organic Enrichment/Low DO," is estimated to be 325 MGD. Operational costs were determined for the 147 POTWs with flows greater than 0.1 MGD. The total flow for these 147 POTWs is estimated to be 324 MGD, or 99 percent of the 184 active POTWs identified in our dataset.

2.3 WQBEL Implementation in Current Climate (BL-WQBEL)

To estimate the change in treatment costs attributable to climate change, two steps in incremental treatment were characterized. First, it was necessary to estimate the incremental treatment beyond TBELs associated with implementation of WQBELs under current or baseline climatic conditions (referred to as BL-WQBEL). Second, it was necessary to estimate the additional change in treatment necessary to meet more stringent WQBELs that account for climate change (*i.e.*, due to anticipated changes in receiving stream flow; referred to as CC-WQBEL).

There is currently very little information on BOD_5 waste load allocations for POTWs, *i.e.*, BL-WQBELs. Although many reaches have been listed as impaired, many jurisdictions have yet to implement mitigation measures, including those that result in more stringent standards for point sources. The effort needed to achieve reductions in BOD to meet a WQBEL will vary based on considerations such as streamflow, particularly low flow 7Q10, effluent loading, and the ambient quality of receiving waters, as well as specific wasteload allocation approaches used by individual jurisdictions. Experience gained from compliance with existing TMDLs can provide some guidance.

A final report entitled *Total Maximum Daily Loads for the Middle Cuyahoga River*, published by the Ohio Environmental Protection Agency (OH EPA) in March of 2000, examines TMDLs for several pollutants related to DO and the subsequent effluent changes for POTWs (OH EPA 2000). The Middle Cuyahoga River is impaired by low DO concentrations due to organic and nutrient enrichment and hydrologic modifications that reduce flow. The primary point source contributors are 6 POTWs on the reach, with flows ranging from 0.004 to 5 MGD (OH EPA 2000). The report details two scenarios for TMDL implementation: current hydrologic modifications are adjusted to increase stream flow, or current hydrologic modifications remain in place. The hydrologic modifications include the removal of two small impoundments to restore a more natural flow regime and a minimum flow release of 3.5 MGD from an upstream reservoir. Currently, the reservoir can reduce flow to zero during dry periods.

The calculated TMDL under the first scenario for $CBOD_5$ is 735 kg/d; under the second scenario it is 411 kg/d.[6] The average projected $CBOD_5$ effluent loading reductions for the 6 POTWs under the first scenario are 13 percent, and 50 percent for the second scenario. Table 2 provides details in terms of POTWs and existing BOD_5 effluent limits and potential reductions under the two TMDL scenarios. (OH EPA 2000). This report not only provides an example for high and low bounds, it also demonstrates the importance of stream flow on the WQBEL.

Table 2. Example TMDL-Based Reductions in POTW $CBOD_5$ Effluent Concentration.

Middle Cuyahoga River					
POTW	**Existing Permit[1] (mg/L $CBOD_5$)**	**Scenario 1 (maintain critical low flow) Limit (mg/L $CBOD_5$)**	**Percent Reduction**	**Scenario 2 (no hydrologic changes) Limit (mg/L $CBOD_5$)**	**Percent Reduction**
Fishcreek	10	10	0	5	50
Kent	10	10	0	5	50
Ravenna	10	8	20	5	50
Franklin	10	8	20	5	50
Akron	10	8	20	5	50
Twin lakes	10	8	20	5	50
Average			*13*		*50*
Christina River Basin					
POTW	**Existing Permit[1] (mg/L $CBOD_5$)**	**Level 1 Limit (mg/L $CBOD_5$)**	**Percent Reduction**	**Level 2 Limit (mg/L $CBOD_5$)**	**Percent Reduction**
Broad Run	25	25	0	22.95	8
PA American	15	12.3	18	11.07	26
Downingtown	10	8.9	11	6.38	36
Kennett	25	17.5	30	16.63	34
Meadowview[2]	22	22	0	22	0
Average			*12*		*21*

[1] 30-day average effluent concentration under the existing NPDES permit.
[2] These are in BOD_5. BOD_5 values can be slightly higher than $CBOD_5$ values because nitrogenous sources are included.

In a second example, EPA (US EPA 2002a) describes the TMDL process for the Christina River Basin, located in Pennsylvania, Delaware, and Maryland. The Christina River is impaired by low DO concentrations and nutrient loading. The report details two TMDL allocation scenarios – “level 1” and “level 2” – involving 5 POTWs discharging to reaches within this basin. The average projected effluent loading reductions for the 5 POTWs under the level 1 scenario are 12%, and for the level 2 scenario the average is 30%.

[6] $CBOD_5$ is *carbonaceous* biochemical oxygen demand, as distinct from total biochemical oxygen demand (BOD), which includes both carbonaceous and nitrogenous oxygen demand. BOD and CBOD are the same if there is no degradable nitrogen in the sample analyzed.

These examples suggest that there is a wide range in terms of incremental change in efficiency, beyond TBELs, required to meet BL-WQBELs. For this study, it was assumed that the likely costs were bracketed by values given in the two reports described above. Low-end and high-end scenarios were evaluated. For the low end load reduction, a BL-WQBEL value of 12% was selected (the average for POTWs in the Christina Basin for the less stringent scenario in that study). A high end BL-WQBEL value of 50% (the reduction required of all POTWs in the Middle Cuyahoga Basin for the more stringent scenario) was selected. Although these appear to be reasonable for a screening analysis, it is important to note that actual effects of WQBEL implementation at individual POTWs will vary greatly and may be beyond this range.

2.4 WQBEL Implementation in Future Climate (CC-WQBEL)

2.4.1 Impact on Low Flows

Uncertainty remains concerning the specific effects of climate change on streamflow in the GLR, however, several factors suggest that low-flow episodes are likely to decrease (US GCRP 2000). Because 7Q10 flows are based on low flow periods, an increase in the number of these periods will result in a reduction of the 7Q10 flow value for a given reach. Given uncertainty in local scale climate predictions, it is difficult to predict the effects of climate change on flows during low-flow conditions. In this study we therefore adopted a bounding analysis framework, by developing low and high end estimates of the effect of climate change on 7Q10 was developed.

Schoen et al. (2006) found that despite uncertainty in the prediction of low flow, in the Mid-Atlantic low flows are likely to get lower as a result of climate change even though the annual hydrograph is not driven by the snowmelt dynamics and the area is predicted to see increases in precipitation. The study used a stepwise linear regression for predicting the future low-flow statistic 7Q10. Based on four general circulation models, Schoen et al. (2006) found a decrease in the 7Q10 over the 21st century and a corresponding need to reduce contaminant load in the future to meet current water quality standards. In the earliest future time period, the predictions of the four models indicate that the ratio of future to current TMDL could range from 0.8 to 1.0, or a reduction of 20% to 0%.

More specific to the GLR, a general characterization of impacts of climate change in the region estimated that annual freshwater flows into the Great Lakes from streams and rivers could decrease by 20% (US EPA 2001). This value was used as the low end estimate of potential changes in 7Q10 in the screening analysis.

The potential impacts of climate change on 7Q10 in the Sangamon River upstream of Monticello, Illinois was investigated by Eheart et al. (1999). Four climate scenarios were used to develop 99 years of synthetic weather data. The 4 scenarios evaluated were: 1) no change in precipitation characteristics; 2) a reduction in the mean precipitation of 25%; 3) a doubling of the standard deviation of precipitation; 4) a combination of scenarios 2 and 3. Results show clearly that for the scenarios assessed, climate change is likely to have a significant effect on low-flow conditions. The 7Q10 event in this basin decreased 63% in response to a 25% drop in precipitation; decreased 57% in response to a doubling of the standard deviation of precipitation; and decreased 84% in response to a 25% drop in precipitation and a doubling of the standard deviation. Because it is not clear whether precipitation will decrease, the scenarios involving a 25% decrement in annual precipitation were regarded as being overly conservative for this analysis. The doubling in the standard deviation in precipitation is quite plausible, however, so 57% was chosen as the high end value for the bounding analysis.

2.4.2 Proportional Impact on Effluent Limit

The bounding analysis developed for this report assumes that there is a direct linear relationship between streamflow and the BOD_5 WQBEL so that a 20% reduction in 7Q10 flow will require a 20% reduction in loading to maintain the same level of water quality. It is also assumed that the reduction in basin-wide TMDL translates directly into a proportional reduction in BOD_5 WQBEL from POTWs.

2.4.3 Incremental Treatment Cost Analysis

As previously discussed, the bounding analysis was based on two scenarios each for BL-WQBEL and CC-WQBEL, i.e., each having a low end and high end value. The four values are shown in Table 3.

Table 3. Bounding Analysis Values.

	Low End	High End
BL-WQBEL (% with respect to pre-TMDL case, TBEL)	12%	50%
CC-WQBEL (% with respect to current 7Q10)	20%	57%

3. Results and Discussion

3.1 Impacts of Climate Change on Costs to Achieve WQBELs

The baseline case removal efficiency for the water quality-based effluent limit was calculated assuming that the pre-WQBEL baseline was the current national average level of BOD_5 removal efficiency, i.e., 85% (US EPA 1998b). For example, if a 50% reduction in BOD_5 loading was required to meet a WQBEL, a POTW currently operating at 85% removal efficiency would need to increase its efficiency to 92.5% to meet the new effluent limits (*i.e.*, a 50% reduction in the 15% that is currently discharged would require removal of an additional 7.5%, so total efficiency would be 92.5%). Furthermore, if an additional 20% reduction was necessary due to a reduction in the 7Q10 low flow event resulting from climate change (to meet the CC-WQBEL) this would mean that a POTW operating at 92.5% removal would have to increase that to 94% to meet the new effluent limits (*i.e.*, a 20% reduction in the 7.5% that is then discharged would require removal of an additional 1.5%, so total efficiency would be 94%). The cost model estimates the cost for each POTW, based on removal efficiency and flow, and sums across all 147 POTWs for each scenario.

Table 4 and Table 5 show the incremental cost results for the 147 POTWs in the GLR. The cost results are reported with respect to a TBEL baseline, as well as incremental to the BL-WQBEL. In the TBEL baseline, annual POTW treatment costs are about $345 million. If WQBELs were implemented, costs are estimated to increase by $14 million to $59 million per year, or an average of $95,000 to $400,000 per facility per year (for the low end and high end, respectively).

Table 4. Estimates of Annual Cost Summed Across All 147 POTWs to Meet Low End BL-WQBEL, with Low End and High End Design Flow Reductions for CC-WQBELs.

Note: "wrt" = "with respect to."

	Percent Change in Load (BL-WQBEL is wrt TBEL; CC-WQBELs are wrt BL-WQBEL)	Percent Removal Efficiency	Annual Cost ($million)	Incremental Annual Cost wrt TBEL ($ million)	Percent Cost Increase wrt TBEL	Incremental Annual Cost wrt BL-WQBEL ($ million)	Percent Cost Increase wrt BL-WQBEL
Pre TMDL case: (TBEL	0%	85.0%	$345	NA	NA	NA	NA
BL-WQBEL with TMDL Load Reduction (low end)	12%	86.8%	$359	$14	4%	NA	NA
CC-WQBEL, Design Flow Low End: with TMDL Load Reduction	20%	89.4%	$380	$36	10%	$21	6%
CC-WQBEL, Design Flow High End: with TMDL Load Reduction	57%	94.3%	$414	$69	20%	$55	15%

Table 5. Estimates of Annual Cost Summed Across All 147 POTWs to Meet High End BL-WQBEL, with Low End and High End Design Flow Reductions for CC-WQBELs

	Percent Change in Load (BL-WQBEL is wrt TBEL; CC-WQBELs are wrt BL-WQBEL)	Percent Removal Efficiency	Annual Cost ($million)	Incremental Annual Cost wrt TBEL ($ million)	Percent Cost Increase wrt TBEL	Incremental Annual Cost wrt BL-WQBEL ($ million)	Percent Cost Increase wrt BL-WQBEL
Pre TMDL case: Technology-Based Effluent Limit (TBEL)	0.0%	85.0%	$345	NA	NA	NA	NA
BL-WQBEL with TMDL Load Reduction (high end)	50.0%	92.5%	$404	$59	17%	NA	NA
CC-WQBEL, Design Flow Low End: with TMDL Load Reduction	20.0%	94.0%	$412	$68	20%	$8	2%
CC-WQBEL, Design Flow High End: with TMDL Load Reduction	57.0%	96.8%	$501	$156	45%	$97	24%

wrt = with respect to.

Estimates of the incremental annual costs of complying with tighter effluent limits expressed as the sum across all 147 POTWs and per POTW are summarized in Table 6. These values suggest that reductions in the 7Q10 flows would add significantly to the costs of WQBEL implementation. The cost results are driven primarily by the increment in treatment efficiency that is required in each scenario and the marginal cost in that region of the cost curve. They also reflect the fact that treatment improvements have a higher unit cost at the high end of the range than the low end (*e.g.*, it costs more to move from 94% to 96% efficiency than to move from 85% to 87%).

Table 6. Summary of Estimated Incremental (with respect to BL-WQBEL) Annual Costs for CC-WQBEL.

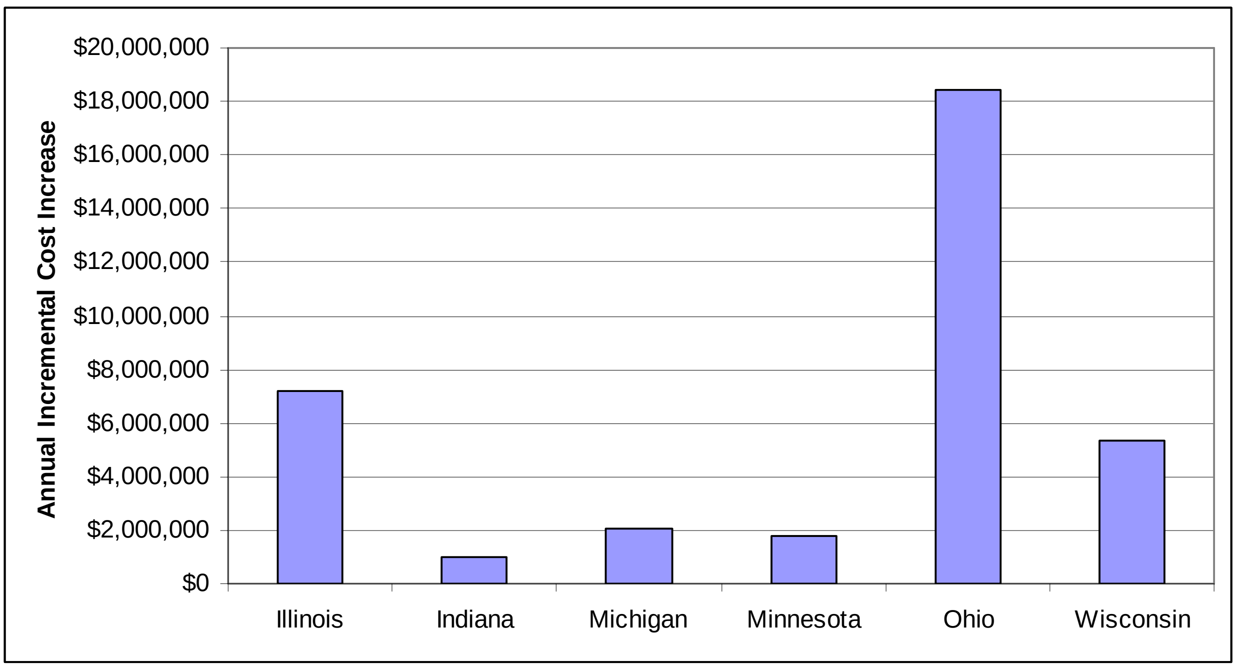
	Low End CC-WQBEL (20% beyond BL-WQBEL)	**Low End CC-WQBEL (20% beyond BL-WQBEL)**	**High End CC-WQBEL (57% beyond BL-WQBEL)**	**High End CC-WQBEL (57% beyond BL-WQBEL)**
	Sum Total Cost:	**Average Cost per POTW:**	**Sum Total Cost:**	**Average Cost per POTW:**
Low End BL-WQBEL (12% beyond TBEL)	$21 million	0.14 million	$55 million	0.37 million
High End BL-WQBEL (50% beyond TBEL)	$8 million	0.05 million	$97 million	0.66 million

The incremental costs associated with changes in design flow appear significant both in absolute terms (millions of dollars) and relative terms (percentage of TBEL costs). In the scenarios assessed here, implementation of WQBELs under current climatic conditions would increase the annual costs POTWs in the GLR region by about 4 to 17% over TBEL levels. Accounting for climate change (i.e. due to reduced 7Q10 streamflow in receiving waters) would increase the incremental cost of implementing WQBELs by an additional 2 to 24% over the current cost of implementing WQBELs.

3.2 State Distribution

Figure 3 shows that the distribution of costs is quite uneven across the states, using as an example the scenario with low end Baseline Compliance and low end Compliance with Climate Change. State costs are proportional to the number of POTWs in impaired reaches and their flow. Ohio, which has made considerable progress in listing reaches, has about half of the POTWs in DO-impaired reaches, and incurs about half of the costs under this analysis.

Figure 3. Distribution of Costs for All 147 POTWs by State: Low end Baseline Compliance, Low end Compliance with Climate Change



3.3 Study Limitations and Future Research

3.3.1 Limitations

There are several limitations to consider when interpreting the results of this research. These limitations are related to the analytic framework and the data. These limitations, along with the screening-level findings, point to the potential for future research in this area.

- *Limited POTW Selection* – This analysis only considers active POTWs on 303(d) listed reaches in the GLR, and should be considered a screening level analysis. As a result, this analysis looks only at a limited number of POTWs in a single region. Climate change would likely impact more POTWs in the region as the reduction in low flows in general (and 7Q10 flows in particular) would occur on all reaches, leading to more reaches being listed as impaired and making additional POTWs subject to WQBELs. More significantly, costs would be much higher if all sources, all impaired reaches, and all pollutants were considered.

- *Load Allocation Assumptions* – A proportional distribution of load reductions was assumed across all BOD_5 sources in an average reach. There will undoubtedly be great variation in actual load allocations and resulting impacts on POTWs. Point source load allocations for POTWS are likely to vary widely due to various considerations such as treatment costs and load allocations for non-point source loads (*e.g.,* agricultural or urban runoff). As a result, the costs associated with meeting each stepwise increment in effluent standards for a given POTW may not reflect what would likely occur. However, from the regional screening perspective of this study, the proportional distribution of load reductions is a reasonable assumption.

- *Cost Curve Assumptions* – The cost curve integrates two unique sets of municipal wastewater treatment cost functions created by the National Research Council (1993) and Qasim (1999). These cost functions are assumed to apply uniformly to all POTWs evaluated. The actual costs of increasing removal efficiency will vary from plant to plant based on existing POTW infrastructure and many complex and site-specific factors. However, there is reasonably good agreement between the NRC cost estimate and the cost elements from Qasim, which provides a level of independent confirmation of their accuracy.

  Another key assumption in the application of the cost curve is that the curve itself is continuous, and an increase in required BOD_5 removal efficiency results in a simple incremental shift up the curve to a new cost point. In reality, there is often a step-wise function of cost relative to treatment efficiency. An upgrade in treatment capacity would require a step up in cost and treatment efficiency (*i.e.,* the nutrient removal, high lime, or filtration system is either there or it is not). The assumption applied in this paper may result in conservative cost estimates as the "full step" up to the costs of the upgrade to increase removal efficiency is not applied, and only the incremental increase in cost is considered. The use of a continuous cost curve provides a simplifying approach to estimating costs and also relates to the proportional waste load allocation assumption previously discussed. If a small POTW is currently operating at the maximum treatment efficiency, and an increase in BOD_5 removal efficiency would require a great step up in costs, then it is less likely to have a stringent TMDL allocation because more cost-effective sources would be targeted.

3.3.2 Future Research

The results of this screening level analysis indicate that there are considerable costs associated with improving POTW infrastructure to meet more stringent WQBELs under future climate change scenarios. The following are several opportunities to improve the accuracy and validity of the estimates developed in this study.

- *Land Use Change* – Future population growth and development along these streams and rivers will result in increased BOD_5 loading as these POTWs are expanded or new ones are built. In fact, it is estimated that by 2025, national BOD loading will return to levels seen before the enactment of the CWA even with continued increases in average treatment efficiency (US EPA 1998b). This increase in overall loading will force even stricter WQBELs and greater focus on non-point sources as treatment efficiencies for point sources such as POTWs are pushed to their technical and economic limits. It would be interesting to investigate how population growth and land use change would interact with the drivers considered in this study. Furthermore, examining the impact of potential changes in hydrology due to urbanization (e.g., the impact of increased temperature, impervious cover on low flow events) and the potential impact Smart Growth programs might have on the future of POTW systems might be illuminating in terms of developing the most cost-effective path to long-term water quality improvement.

- *Uncertainty Analysis* – This screening analysis is formulated as a bounding problem, with high and low endpoints for two different system characteristics (i.e., implementation of baseline WQBELs and effect of climate change on design flows). It would be straightforward to describe these parameters as distributions, rather than simply point values, and then carry out a simple Monte Carlo-type analysis to create a phase space of outcomes. Such an analysis could be conducted in concert with expert elicitation on several key inputs, such as the incremental stringency of WQBELs (compared to TBELs), the shape of the cost curve beyond TBEL, and the effect of climate change on receiving stream flow.

- *Climate Change Impact on Low Flow* – There has recently been additional research investigating the potential impacts of climate change on low-flow conditions (e.g., (Nelson et al. 2006). This research could contribute to a more accurate characterization of the impact of climate change on 7Q10 flows.

- *WQBELs in Unimpaired Reaches* – This study focused on implementation of WQBELs in impaired reaches, where TMDLs will need to be developed. However, WQBELs may also be imposed by permit writers in reaches that are not on the CWA 303(d) Impaired Waters list, within essentially the same regulatory framework (and with potentially similar effects of climate change on compliance costs) as described here. It would be useful to determine how often WQBELs are currently (and potentially) implemented in unimpaired reaches, and to adjust the cost estimates accordingly.

- *Other Pollutants* – The focus of this assessment was limited to those impairment types related to DO levels. Expanding the scope beyond DO-related impairments would increase the number of reaches – and thereby POTWs – included in the assessment, giving a more comprehensive view of potential impacts of climate change on these systems.

- *Additional Pollution Sources* – Effluent from POTWs is only one source of pollution that impairs waterbodies. There are many industrial and agricultural sectors that discharge significant pollutant loads. WQBELs will also affect the stringency of discharge permits for these facilities. Expanding the scope to include additional source categories would require creating additional cost curves for each sector evaluated.

- *Expanding the Geographic Scope* – This study focused only on POTWs in impaired reaches in the GLR. From the standpoint of more accurately estimating national costs, it would be useful to develop a national-scale estimate of compliance costs.

4. Conclusions

Climate changes such as warming temperatures, "flashier" precipitation events, and lower low flows will have significant impacts on water resources. Changes in these conditions have direct implications for decisions water resource managers are making today, because long-term investments, such as municipal wastewater treatment infrastructure, are expected to perform for decades into the future. The effectiveness of these infrastructure investments will, in part, be determined by their performance under future climate conditions that differ significantly from conditions observed over the past century.

As a general screening analysis, the results of this study suggest that climate change could have a significant effect on two of the most important water quality programs in the United States: attaining water quality standards (through WQBELs in general and the TMDL program in particular) and POTW financing. As the TMDL program grows and more impaired reaches are listed, thousands of additional sources may be included. Furthermore, as required treatment efficiencies become more rigorous, the cost of treatment grows dramatically, so that even small increments associated with climate change-related reductions in low-flows may have significant costs. This analysis, like that of Eheart et al. (1999) and Schoen et al. (2006), suggests that water resource planners should account for the possibility that climate change will affect design flows in impaired water bodies given that any change in future climate that results in significant reduction in flows would affect assumptions made in WQBEL calculations, with negative consequences for in-stream water quality.

As for POTW financing, given that the design lifetime of POTWs is many decades – long enough for the effects of climate change to be manifested – this analysis also suggests that long-range planning for the State Revolving Fund (Clean Water Act Title VI) and other POTW financing mechanisms should be aware of the possibility that treatment efficiencies required to meet WQBELs will need to be more stringent than previously recognized. Thus, costs of treatment would be higher. The Clean Water Needs Survey (CWNS) released in 2004 indicates that $57 billion will be required for improving wastewater treatment systems in the U.S. over the next 20 years (US EPA 2003). The results of this study indicate that the costs of adapting to climate change, in addition to meeting the requirements of future WQBEL implementation, could be considerable.

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