Water quality modelling, risk analysis and decision-making: an integrated study

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A thesis in the Department of
Building, Civil & Environmental Engineering

Presented in partial fulfilment of the requirements for the degree of
Master of Applied Science (Civil Engineering)

Concordia University
Montreal, Canada

March 2012

CONCORDIA UNIVERSITY

School of Graduate Studies

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Master of Applied Science (Civil Engineering)

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ABSTRACT

Water quality modelling, risk analysis and decision-making: an integrated study

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Concordia University, 2012

Falling detection limits, the proliferation of chemical contaminants and the rising population densities of the world's watersheds are erasing the traditional conception of wastewater and drinking water as demarcated academic disciplines. This increased interconnectedness reflects increased awareness of the extent to which human activities impact water resources and to what extent these impacts are felt downstream. This text integrates the candidate's contributions to the fields of industrial-scale wastewater treatment, municipal-scale wastewater management planning, and drinking water management and regulation made during his postgraduate studies. The collected works constitute a novel civil engineering dissertation that challenges the historic compartmentalization of subfields in water resources management.

1) A new type of wastewater treatment system is modelled with traditional computational fluid dynamics ("CFD") methods in order to optimize operating conditions and design features and to model its principal hydrodynamic characteristics.

2) Risk analysis techniques are used to build a decision model for municipal-scale management of wastewater discharged to highly transient water bodies. A risk analysis framework uses environmental and economic impact valuations to translate CFD model output into concrete decisions about infrastructure investment.

3) The claim that improved analytical ability leads to stricter regulations than justified by available toxicological data is evaluated with respect to drinking water. The relationships between historical and present maximum contaminant levels and goals are evaluated, and the overall drinking water regulatory apparatus is outlined. Analytical ability is demonstrated to have a weak effect on resulting regulation, and the tightening of regulations is shown to be unlikely in the absence of updates to the underlying toxicological model.

4) Drinking water regulatory structures are compared with respect to their ability to protect public health in light of several widely reported outbreaks in Canada since 2000. Claims of inadequate government intervention and proposed alternative regulatory arrangements are contextualized using principles of risk perception and evaluated using principles of risk analysis.

5) Drinking water regulation in the United States is deconstructed to understand to what extent the landscape of rules and standards reflects the balancing of risks with the costs and benefits of treatment. The research points to risk biases that make low-risk, high-occurrence contaminants more likely to be regulated than high-risk, low-occurrence contaminants. Decentralization along the lines of the Canadian model is shown to have the potential to improve water quality outcomes in a way that is consistent with risk valuations already established.

ACKNOWLEDGEMENTS

My advisors, Samuel Li and Ketra Schmitt, have been invaluable sources of guidance on the research presented in this thesis and in respect of my professional endeavours in general since my time as an undergraduate student. Laleh Yerushalmi has also been very generous with her time and careful analysis of the first chapter of my thesis.

I am grateful for the funding I have received throughout my master's degree. My research has been supported by graduate scholarships provided by the Power Corporation of Canada (awarded by the School of Graduate Studies) and the Fondation Universitaire Pierre-Arbour as well as by the generous funding allocated to me by my advisors in the Department of Building, Civil & Environmental Engineering and the Centre for Engineering in Society. I am determined to make good on their investment.

Thanks also to my friends and colleagues at Conestoga-Rovers & Associates for their support and encouragement throughout my master's studies. I have benefited tremendously from the opportunities CRA has given me to refine my expertise in a practical environment. They have made an engineer out of me.

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INTRODUCTION

One man's wastewater is another man's drinking water – almost. Although the hydrologic cycle tends to dilute wastewater so as to physically and conceptually differentiate it from drinking water, the demarcation between the two as distinct disciplines of research and management is disappearing.

Concepts of drinking water treatment and wastewater discharge criteria are longstanding acknowledgements of the link between the two. However, this connection is largely implicit, and water resources management has traditionally been characterized by fragmentation between actors in drinking water procurement and treatment on the one hand and wastewater treatment and discharge on the other. Indeed, recognition of this fragmentation has led to the coalescence of the 'Integrated Water Resources Management' (IWRM) paradigm to improve coordination of land-use, wastewater and drinking water planning in an age of overstressed water resources (UN WWAP 2009).

The IWRM has tended to frame thinking about protecting water resources in the developing world (e.g. Swatuk 2005; Goldin et al. 2008) where drinking water has become a primary vector for pathogens leading to short-term illness and death. However, the economic, social, technical, legislative and geographic barriers to adoption of an integrated strategy to water resources planning (Thomas and Durham 2003) have also been examined in the contexts of Canada (Creighton 1999; Roy et al. 2009; Shrubsole 2004) and the United States (Ballweber 2006; Stakhiv 2003). Although the risks to drinking water supplies tend to be less dire in industrialized countries, understanding and overcoming the barriers to an integrated approach to water management is required in order to respond to the public's expectations of water quality.

Indeed, a number of well-publicized outbreaks in Canadian public water systems since 2000 have reminded the public that industrialized countries are not immune to the public health and environmental issues most commonly associated with the developing world. Additionally, a universe of thousands of chemical drinking water contaminants owing almost entirely to the discharge of industrial effluents has been inventoried since the 1970s. These contaminants largely represent increased risks of long-term illness, notably, different cancers.

Improved analytical abilities now allow for routine detection of chemical contaminants in levels on the order of one part per billion. Over the past generation, we have become aware of widespread but vanishingly small concentrations of chemical contaminants in drinking water. Unfortunately, knowledge of dose-response profiles at these low levels is highly uncertain and poses a dilemma for decision-makers by making precautionary regulation increasingly demanding yet decreasingly likely to result in material protection of public health.

This represents a huge puzzle for public health decision-makers trying to optimize the allocation of finite resources to minimize incongruent (short-term versus long-term) and uncertain (long-term) public health risks. These challenges are perpetuated by a policy feedback loop: increased understanding of drinking water contamination often leads to a policy response, while regulations and policies are themselves instrumental in setting scientific research priorities (Schmitt and Calder 2011). A long-term vision to reconcile drinking water quality expectations and outcomes is a crucial dimension to an integrated approach to water resources management in the developed world. Much as in the developing world, developed countries have an emerging need for an integrated approach to water resources management. Whereas in the developing world, this is driven by immediate threats to human health, in the developed world, it is to ensure long-term efficiency in decision-making and fair reallocation of the burden of water treatment and resources protection from downstream consumers to upstream emitters.

Environmental regulation in developed countries is criticised variously for not being sufficiently protective of public health and, conversely, for imposing stricter standards than are justified by available data. Wastewater and drinking water are not exceptional in this regard. However, they are distinct in being the only spheres of environmental management to require policy and technical intervention both by polluters and consumers: as we are coming to realize, one man’s wastewater is another man’s drinking water.

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The changing landscape of environmental management is especially relevant for engineers, who will be called upon to build, design and maintain the physical manifestations of an integrated approach to water management. It is therefore no longer sufficient for engineering decision-makers and managers to have a technical background limited to either drinking water or wastewater management; increased integration demands from tomorrow’s decision-makers an expertise that straddles the extant divisions of today.

This thesis is the culmination of the candidate’s civil engineering research. Each chapter is an academic journal article either published, submitted for publication or in the final stages of preparation. The chapters represent novel research into different parts of the drinking water/wastewater continuum: small-scale wastewater treatment modelling, large-scale wastewater management decision-making and drinking water management decision-making. The chapters stand alone as independent paths of inquiry but, together, constitute a research program that has grounded the candidate in the wide foundations that underlie an integrated approach to water resources management.

The progression of the five chapters of this thesis reflects the directionality of water as a medium that links human populations: micro-scale wastewater treatment, followed by macro-scale sewage plume management upstream, then drinking water analysis and management downstream. It also provides continuity in terms of quantitative methodologies and a steady upward progression in the weight of qualitative discussion: the thesis opens with a traditional CFD study that is followed by the development of a framework to nest CFD within a risk-based decision model and concludes with risk-based policy analyses.

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Chapter 1 describes a computational fluid dynamics (CFD) model developed for a new type of wastewater treatment bioreactor for which a working lab-scale prototype exists. The model is first run under the conditions applied to the laboratory-scale prototype. The CFD output is compared to the laboratory output to validate the CFD model. Then, the model is modified variously in terms of operating conditions and geometry to study the effects of these changes on the flow pattern and operating characteristics and to thereby find the optimal setup. In addition to being vastly less resource-intensive than performing an equivalent number of tests on the laboratory-scale prototype, the CFD model yielded valuable information on the magnitude and location of turbulent eddies within the bioreactor. This work has been submitted to the Journal of Environmental Engineering as of the compilation of this thesis.

Chapter 2 demonstrates how CFD models of transient, uncertain or probabilistic systems can be nested within a risk analysis framework to guide decision-making. A model for large-scale wastewater treatment decision-making is proposed for the highly transient environments of tidal waters. Two-dimensional numerical models of sewage plumes in such environments illustrate the high sensitivity of water quality to input parameters such as seasonal stratification and tidal condition that vary continuously with time. The model outlines how probabilistic methods would translate discrete snapshots of water quality under time-variable conditions into a quantitative, constant description of water quality that can be used to evaluate the costs of ecological and economic impacts. This research was been presented at the Society for Risk Analysis annual conference in 2011 and won the candidate the Student Merit Award in the Ecological Risk Assessment Specialty Group. It is in the final stages before submission to a journal.

Chapter 3 is a quantitative policy analysis of the effects on improved analytical ability on drinking water regulation in the United States. It is the first study on the relative importance of detection ability, toxicological knowledge and cost factors in drinking water regulatory decision-making. This study was carried out to study claims from industry that better detection limits lead to regulation stricter than what is justified by toxicological data. We find that drinking water regulation is in fact driven primarily by improved knowledge of the dose-response profiles of contaminants and not by improved detection limits. However, improved detection ability is demonstrated to create more targets for regulatory analysis which can subsequently lead to regulation. This work has been published in Environmental Science & Technology (Calder and Schmitt 2010).

Drinking water quality management in Canada is based primarily on provincial/territorial intervention. This contrasts with the American and European approach of uniform, legally enforced regulation at the federal or super-federal level. The Canadian government has been widely criticized for not giving its drinking water quality guidelines legal effect. **Chapter 4** investigates the principal arguments for legally binding federal drinking water guidelines for chemical contaminants, compares popular claims about water quality with available data and adds context with perspectives on risk perception and general principles in drinking water management. The work compares current and proposed regulatory frameworks in terms of their implications for protectiveness of public health, responsiveness to emerging contaminants and equity among populations. It shows that increasing the responsibility of the federal government could potentially restrict its ability to adopt health-protective drinking water quality standards, slow its reaction to emerging contaminants and undermine its goal to protect public health. This is contrasted with an analysis of the practical difficulties inherent in sub-national water regulation. The work concludes by identifying opportunities for data collection that would allow for more confident, definitive pronouncements on risks and benefits of different regulatory structures.

Chapter 5 is an analysis of drinking water regulatory decisions made in the United States since the passage of the Safe Drinking Water Act. Two major shifts in the realm of drinking water regulation in the United States are shown to have occurred that, together, jeopardize the ability of centralized regulation to protect public health. The first is a shift in regulatory attention from contaminants relatively widespread and common to many urban areas to those that occur infrequently in a small number of water systems. The second is the entrenchment of the Environmental Protection Agency's (EPA) burden to demonstrate regulatory benefits on a national as opposed to a regional level for any proposed drinking water regulation. Continued protection of public health to a level established by past regulation depends on an acknowledgement by decision-makers of the challenges posed by the current centralized regulatory framework and the adoption of measures to bring regulatory decision-making into line with the nature of today's contaminants and the public's expectations of the distribution of risks and benefits. This work is in the final stages of editing before submission to a journal.

LITERATURE REVIEW

The research program compiled in this thesis was carried out to ground the candidate in the broad foundations underlying an integrated approach to water resources management. Each of the five chapters of this thesis stands alone as a separate research question aimed at audiences in established wastewater and drinking management audiences. Chapters 3 to 5, on drinking water, however directly address the theme of developing a long-term vision for water quality expectations, outcomes and management.

Literature reviews are provided for each chapter separately. The reviews presented in this section summarize the work that has been done in and around areas of inquiry of each chapter, identifying primary sources and references, as well as the contribution of each chapter to its respective research community. This section is not an exhaustive overview of every reference cited by each chapter.

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Chapter 1: Computational fluid dynamics model of a BioCAST multi-environment air-lift bioreactor

The wastewater treatment technology modelled in this research has undergone laboratory trials that have produced data on its contaminant removal performance (Yerushalmi et al. 2011) and principal hydrodynamic characteristics (Behzadian 2010). There is a substantial body of literature on the use of computational fluid dynamics (CFD) to study air-lift bioreactors similar to, though usually simpler than, the one modelled in this research. This literature was valuable in providing other researchers' perspectives on the aptness of different turbulence models to describe the physics of bioreactors and the relative importance of different physical characteristics. The majority of researchers (e.g. Van Baten et al. 2003; Mudde & Van Den Akker 2001) have used a two-phase (air/water) k-ε turbulence model. Some (e.g. Tabib et al. 2008) have used large eddy simulation (LES). Still others (e.g. Volker & Hempel 2002; Heijen et al. 1997) expanded the model to account for a dispersed solid phase. All of the work surveyed considered the flow in the bioreactor to be driven exclusively by the dispersed gas phase. A review of the literature found no hydrodynamic study of a wastewater reactor with a clarifier isolated from the circulating flow.

Manuals and guidelines on best practices for CFD modelling were instrumental in creating the model (Chen & Jaw 1998; Ranade 2002; ANSYS 2009). The chemical engineering literature provided semi-empirical formulae and guidelines to facilitate making assumptions on various physical concepts such as bubble diameter and flow regimes that defined the physics input of the model (Akita & Yoshida 1974; Hills 1976; Levich 1962).

The model output was validated with respect to values reported in laboratory trials (Behzadian 2010) and standard estimates available in the literature for similar geometries (Bello et al. 1984; Chisti et al. 1988).

Chapter 2: A probabilistic decision-making framework for the management of sewage plumes in a tidal environment

Hydrodynamic models have recently been developed to plot sewage plumes in tidal environments and illustrate the high sensitivity of the plume shape and dilution to time-variable input parameters (e.g. Li & Hodgins 2010; Liu et al. 2008). This research provides a model for decision-making in such fundamentally probabilistic, extremely transient water systems. Our task is to understand how the discrete, representative model outputs provided by CFD research can be aggregated to describe water quality on average.

A review of the literature reveals that decision analysis in water quality management has generally been confined to the development of strategies either to account for uncertainty of input parameters (sewage loading, wind speed, etc.) or the uncertainty of models used to translate physical predictions into ecological and economic impacts (Beck 1987; Bingham et al. 1995; Reckhow 1994). Decision analysis does not seem to have found a place structuring water resources decision-making in contexts that are not only or necessarily characterized by uncertainty but also by fundamental variability as is the case in tidal waters.

This gap is perhaps due to the relatively recent accessibility of computing power that allows for a single study to comprise dozens of CFD simulations. Using traditional CFD methodology with point estimates for the input variables, a relatively large number of simulations are required to fully characterize the water body under the different combinations of values possible for the input variables. The gap may also reflect a lack of modelling tools to resolve highly transient water quality into a singular index that may be used for the purposes of impact valuations. Indeed, a review of the literature indicates that the phenomenon of variable water quality has not been explicitly explored. Instead, in cases of variable water quality, semi-arbitrary management decisions tend to prevail with respect to which modelling output (e.g. worst-case, 95th percentile) data are used for decision-making. This in turn may reflect the insensitivity of the dose-response models underlying impact valuations to the time-weighting of ecological and economic impacts.

The framework built in this chapter aims to introduce decision analysis techniques into CFD research to improve its potential for facilitating decision-making in highly transient water bodies.

Chapter 3: Role of Detection Limits in Drinking Water Regulation

This research was inspired by claims by recurring claims by commentators in environmental science and policy that improved detection capabilities precipitate more stringent regulations. This phenomenon is sometimes termed 'the vanishing zero effect' or 'detection limit creep' (Elston 2005). This is the first academic study on the existence of this alleged phenomenon, although two other academic articles addressing it have been identified (Belluck & Benjamin 1993; Elston 2005). More commonly, the concept is mentioned indirectly in research articles assessing risk in regulatory decision-making or discussed at conferences (e.g. Long 2004; Rodricks 2001). At least one article in an industry trade journal predicts that detection limits would be the 'dominating force' of environmental regulations in the 1990s and beyond (Linstedt 1993). In some cases, the articles reviewed suggest that improved detection capability can itself motivate regulation at levels departing from available toxicological data (Elston 2005; Linstedt 1993).

Previous research has been directed at this question from a technical perspective on a case-by-case basis; Ferguson et al. (2007) demonstrate the technical feasibility of lowering the practical quantitation limit (PQL) of arsenic, based on the detection ability afforded by a variety of commercially available technologies. Otherwise, researchers have commented tangentially that the detection limits required by the United States Environmental Protection Agency (US EPA) are in fact below the associated health standards it sets, suggesting that, in some contexts, analytical capability is playing a decreasingly important role (Hamilton et al. 2003). Whatever the relationship to regulation, advances in analytical chemistry are a documented concern to stakeholders in environmental policy; Johns (2007) writes that it is necessary to consider the added costs of more advanced analytical procedures that would be borne by industries and small municipalities in the event that state regulators require them.

Much of the analysis was based on US EPA's two reviews of the technical feasibility of considering lowered detection limits in decision-making (2003, 2009a). Literature on contaminants that are candidates informed analysis on the likelihood of improved analytical abilities precipitating new regulations in the near future (US EPA 2009b). Literature on cost benefit analysis at US EPA (Morgenstern 1994) and drinking-water decision-making (NRC 1999) as well as technical literature on detection limits (Gibbons 1994) was invaluable in outlining which contaminants are regulated, and to what levels.

Chapter 4: Drinking water regulation: risks, benefits and the hunt for equality in the Canadian context

In 2005, the Office of the Auditor General of Canada (OAGC) undertook two audits of federal engagement in drinking water quality, one with respect to Indian Reservations (OAGC 2005a) and one with respect to all other obligations and functions (OAGC

2005b). Most opportunities for improvement highlighted by the audits pertained to ensuring competent operation of small systems, adequate testing and sufficient funding, especially in Indian Reserves. One finding criticised the pace at which the federal Drinking Water Guidelines (that primarily pertain to chemical contaminants) are updated and developed. As of its 2009 report, the OAGC considers the federal government to have 'made satisfactory progress' with respect to the 2005 evaluation and noted the implementation of a process to improve the pace with which chemical contaminants are evaluated (OAGC 2009). A search of the federal government documentation reveals that the vast majority of publications fall into the following categories: outlines of upcoming research needs and identification of priority areas; resources for well-water users; conclusions of environmental assessments; technical information on select contaminants; summaries of international development initiatives; explanatory scientific material for the public; reviews of international commissions and agreements; and resources for agricultural workers.

The environmental geography literature is rich with study and perspectives on overall water governance in Canada (e.g. Bakker 2002; de Loë et al. 2007). This body of literature focuses on source water management and responsibility with respect to the institutional, legal, geographic, economic, social and environmental challenges that face decision-makers. Academic literature on drinking water safety focuses heavily on control of pathogenic contamination and effective operation of municipal treatment facilities (e.g. Hrudey and Hrudey 2004; Hrudey et al 2006; Hrudey 2011; Schuster et al 2005) and source-water protection (e.g. Davies and Mazumder 2003) as opposed to risk management of chemical contaminants or the structure of drinking water regulations across jurisdictions. Policy questions in the context of drinking water are typically confined to the current context of provincial and territorial responsibility (as in McMullan and Eyles 1999). One paper however identifies several potential shortcomings in the regulatory mechanisms intended to ensure water quality from the perspective of environmental law (Pardy 2004). Hill et al. (2008) wrote a paper specifically addressing the lack of literature on Canada's unique, highly decentralized water regulatory model. While this article was not specific to <u>drinking</u> water regulation, the authors inventoried the drinking water management practices of each province and territory and put Canada in the context of the primary scholarly arguments for and against centralization of environmental regulation.

On the other hand, there are not many studies structured as specific policy analyses of the jurisdictional issues involved in drinking water regulation in the Canadian context. This is at least partially due to the fact that shared water governance in Canada is still in its formative stages (Nowlan and Bakker 2010). Notably, a review of the literature did not uncover any discussion on alternative arrangements for administration of standards for chemical contaminants. Although pathogenic contamination of drinking water leads to short-term illness and death and therefore represents a more immediate concern for drinking water decision-makers, the associated standards and quality goals are well established. On the other hand, developing and enforcing standards for chemical contaminants is an extremely complicated and resource-intensive exercise. Chemical contaminants also attract the attention of environmental advocacy groups to a level that often exceeds the interest they direct at safeguarding water from microbiological contamination.

This paper is close in scope to the work of Hill et al. (2008). However, it zooms in on drinking water regulation, risk perception and management and the administration of chemical contaminants. This paper is also unique in explicitly addressing the claims made by leading environmental groups regarding drinking water management.

Chapter 5: Drinking water regulatory decision-making: sensitivity to contaminant occurrence and regulatory philosophy in the United States

The highly centralized drinking water regulatory system in place in the United States is studied to understand how its risk abatement has varied over time, and how the current regulatory setup responds to emerging contaminants. This evolution is put in the context of the changing spatial distribution of contaminants: whereas the first chemical contaminants regulated were common to large swaths of the country, contaminant occurrence in drinking water is becoming increasingly system-specific. The response to this changing landscape by lower levels of government is also evaluated to understand how the apparatus as a whole is adapting to this changing context.

This analysis is facilitated almost entirely by data published since the 1960s by US EPA and its predecessors. A review of the literature found no similar study on the evolution of water resources risk management over time.

CHAPTER 1

Computational fluid dynamics model of a BioCAST multi-environment air-lift bioreactor[1]

A computational model was developed to study the hydrodynamic characteristics of a new multi-environment air-lift bioreactor known as BioCAST. This CFD study considerably expands on the laboratory experiments by exploring the hydrodynamic characteristics of multiple combinations of geometries and operating conditions and by providing a visual illustration of the liquid-phase flow patterns. The model was first tested against preliminary laboratory results to ensure its validity. This included comparing two simplified geometries for the three-disc prototype air sparger assembly in order to determine which led to results closer to laboratory measurements. A torus geometry was found to better represent the prototype than a single disc. The model was then modified to evaluate the hydrodynamic characteristics of alternative operating conditions and physical geometries beyond what would be possible in the laboratory. The flow pattern in the outer clarifier zone was shown to be very sensitive to the geometry of the reactor wall separating the clarifier from the inner microaerophillic zone. Establishing a smooth, upward flow pattern in the clarifier was shown to be possible only when the clarifier was sufficiently shielded from the circulation in the anoxic cone below.

1.1 Introduction

A multi-environment wastewater treatment technology has recently emerged as a technology for high-rate removal of carbonaceous contaminants together with nitrogenous and phosphorous nutrients. The technology is known as BioCAST and consists of two interlinked bioreactors. The first bioreactor, explored in this research, integrates aerobic, microaerophilic and anoxic conditions and is based on conventional airlift designs. This system offers high removal efficiencies of carbon and nitrogen under substantially simpler operation procedures and while producing far less sludge than traditional approaches relying on unit processes in series (Yerushalmi et al. 2011). The liquid circulation within and between the environments is driven by the inflow of air from a sparger at the base of the aerobic zone, located in the centre of the first bioreactor. The microbiological activity in bioreactors is dependent on aeration levels, mixing and shear rates (Ma et al. 2003). These hydrodynamic characteristics are in turn dependent on reactor geometry and operating conditions as investigated by other authors who have developed computational or laboratory-scale models (e.g. Camarasa et al. 2001).

The present research uses computational fluid dynamics (CFD) methods to model the principal hydrodynamic characteristics of the first BioCAST reactor on a laboratory scale

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[1] This chapter is the basis for a journal paper with the same title co-authored by SS Li & L Yerushalmi, submitted to the *Journal of Environmental Engineering* in March 2012.

and explores the effects of variable operating conditions and geometry. This is the first study to use CFD to examine the hydrodynamic characteristics of the BioCAST reactor. CFD allows for greater insight into the particularities of the flow patterns in the reactor and allows for study of many combinations of operating conditions and geometries. The time and cost required to carry out a similar study on the physical prototype would have been prohibitively high. This research first reproduces the geometry and conditions studied in laboratory trials both to validate model outputs against experimental observations and to quantify flow patterns, pressure and gas holdup contours and other hydrodynamic information that could not be evaluated over the entire domain in the laboratory. We then vary reactor and air sparger geometry, wastewater loading and gas inflow rate in order to assess the effects of these parameters on the hydrodynamic characteristics that in turn control reactor performance.

CFX by ANSYS (2010) was used to define reactor geometry, overlay a computational mesh and solve the turbulence model we developed over the flow domain. The post-processing tool was used to visualize the results, generate graphics and produce output files tabulating the spatial distribution of velocity and pressure.

We find good agreement between the results of the computational fluid dynamics (CFD) model and laboratory trials. Significantly, the model demonstrates the difficulty of achieving smooth, upward flow in the outermost clarifier zone, a feature unique to this design. The isolation of this region from the air-driven flow through the inner aerobic and microaerophillic zones translates into a substantially slower current dominated by throughput of wastewater on its way out of the reactor. The flow in the clarifier is easily influenced by the shear of fast, circulating fluid in the anoxic cone below. We therefore demonstrate that geometry and operating conditions must be carefully controlled in order to ensure smooth flow in the clarifier and sedimentation of biological flocs.

The multi-environment BioCAST reactor studied here has an outer anaerobic clarifier that is isolated from the air-driven flow circulating between the interior aerobic and microaerophillic zones. The substantially slower flow in the clarifier therefore has the potential to be influenced by the eddies in the adjacent zones that exert shear forces on the bottom of the clarifier domain and by the wastewater throughput that, under some operating conditions, may have a significant impact on the flow pattern in the clarifier. It is primarily these peculiarities of the outer clarifier that distinguish the model developed here from models of wastewater reactors developed over the past ten to twenty years. These past studies have, in contrast, examined relatively simple internal- and external-loop geometries, typically with two-phase liquid/dispersed gas flow domains. The k-ε turbulence model has been used almost exclusively by CFD researchers, with a minority opting for large eddy simulation (LES) (Tabib et al. 2008). Some studies expand the flow domain to include a dispersed solid phase, e.g. (Volker & Hempel 2002, Heijnen et al. 1997). All of the work surveyed has considered the flow to be driven entirely by air input.

The effectiveness of air-lift systems for wastewater treatment derives from the use of rising gas both for agitation and aeration. The system can be constructed so as to direct flow through variably aerated zones of different biological activity with greater turbulence, less shear stress and lower energy costs than comparable impeller-driven technologies (Vial et al. 2002). The turbulent conditions that allow these operational advantages come at the cost of modelling difficulties, described in further detail below. Considerable research has been directed at turbulence modelling over the past century (Chen & Jaw 1998). This work is in part an example of this effort.

1.2 Geometry and mesh generation

The bioreactor was created in CFX DesignModeler based on the dimensions of the laboratory-scale unit as shown in Figure 1.1. A 90-degree model with two planes of symmetry allowed for design, meshing and computation to be carried out on one quarter of the reactor and extended to the full cylindrical shape. The model is therefore assumed to be symmetrical with respect to the two faces of the 90-degree wedge. Figure 1.2 provides an overview of the 90-degree wedge modelled in CFX. Certain elements of this model were modified in some runs, as described in Section 1.6: Model run and results.

The horizontal spout between the aerobic and microaerophilic zones was modelled as one 3-cm-radius 30-cm-long extrusion breaching the wall, centred at 45 degrees from both planes of symmetry. The outlet from the clarifier is not modelled geometrically but corresponds to the top surface of the clarifier zone.

A small wedge of radius 5 cm was defined in the middle of the top surface of the aerobic zone to allow for two boundary conditions: water inlet through the central wedge and degassing around the wedge on the remaining surface. CFD models developed for reactors of other geometries have tended to omit hydrodynamic effects of throughput, e.g. (Volker & Hempel 2002), likely on account of its much lesser influence on the flow pattern relative to the effect of inflowing air. However, inclusion of wastewater throughput allows us to confirm the effects of varied wastewater flow on the hydrodynamic characteristics of the reactor, particularly in the outermost clarifier zone, which is isolated from the air-driven flow of the aerobic, microaerophilic and anoxic zones. An unstructured grid with an element size of 9 mm was generated over the reactor domain. Including geometric protrusions for these inlet and outlet zones facilitated the definition of boundary conditions. Including them in the model domain appears to have very little effect on model output (Ranade 2002).

The air sparger apparatus used in the laboratory prototype reactor was of complex geometry approximating three rings centred over a solid pump apparatus at the base of the aerobic zone. In the laboratory prototype, air was introduced through 63 circular holes of diameter 1 mm distributed over the surface of the reactor. Because these inlet ports were smaller than the diameter of the bubbles in the fluid and because the two-phase dispersed gas fluid has not been realized at the boundary of the air inlet ports, it was not realistic to model flow at the ports. Therefore, simplified sparger geometries were used: in the first eight runs, the air sparger was modeled as a single 7.6-cm-radius cylinder within the aerobic zone with its base at the bottom of the wall between the aerobic and microaerophilic zones and extending upwards 6 cm. The aerobic and microaerophillic zones therefore communicate through a 1.4-cm-thick circular envelope around the edge of the sparger, down the base of the wall between the two zones. In the remaining 16 runs, the sparger was modelled as a torus of outer radius 6.6 cm and inner radius 2.4 cm (therefore with a circular cross section of radius 2 cm). In this case, the aerobic and microaerophillic zones communicated through the centre of the sparger, a circle of diameter 4 cm, and between the outer edge of the sparger and the base of the microaerophillic zone, a ring of outer radius 9 cm and inner radius 7 cm. Six of the runs with torus sparger geometry corresponded to runs that had used a cylindrical sparger and were conducted to test model sensitivity on the different sparger assumption. The conditions used in all runs are specified in Table 1.1.

1.3 Fluid domain and model physics

The reactor volume was modelled as a two-phase, isothermal mixture of air at 25°C dispersed in liquid water as bubbles with a uniform diameter. This hydrodynamic model essentially corresponds to the homogeneous ‘bubbly flow’ regime described in chemical engineering literature (Ishii & Hibiki 2011). The prevailing flow regime in a bubble column varies between this homogeneous condition and heterogeneous slug and churn turbulent flows as a function of superficial gas velocity, gas rise velocity and column diameter (Shah et al. 1982). However, the present hydrodynamic model does support heterogeneous flow conditions and will in all cases force bubbly flow. Therefore, the model inputs and outputs must be manually validated with respect to the conditions for bubbly flow to ensure that the homogeneous fluid domain model is an appropriate starting point for computing the hydrodynamic conditions of the reactor.

We find in the literature estimates on the limits of bubbly flow in terms of maximum superficial gas velocity ($u_{g,s}$) (Fair 1967): $\mathbf{u}_{\mathbf{g,s}} < 0.05\ m/s$ and maximum and minimum rise velocity ($u_{g,r}$) (Levich 1962): $0.18 < \mathbf{u}_{\mathbf{g,r}} < 0.30\ m/s$. For column diameters greater than 10 cm, bubbly flow is confined to superficial gas velocities lower than 0.05 m s^{-1} and is transitional up to about 0.07 m s^{-1} (Hills 1976).

The superficial and rising gas velocity fields predicted by the hydrodynamic model for different conditions must be compared with these guidelines to ensure the validity of the dispersed gas phase model and hence the model output in general. Experimental trials confirmed that bubbly flow exists at least within the range of 10 to 70 L min^{-1} of gas input (Behzadian 2010).

The bubble diameter (d_b) was estimated based on the correlation proposed by Akita and Yoshida (Akita & Yoshida 1974):

$$d_b = 26 \cdot D_c \left(\frac{D_c g \rho_L}{\sigma} \right)^{-0.5} \left(\frac{g D_c^{\,3} \left| \mathbf{u}_{g,s} \right|}{\nu_L^{\,2} \sqrt{g D_c}} \right)^{-0.12} \tag{1}$$

Where D_C is column diameter, ρ is density, ν is molecular kinematic viscosity and subscripts L and g denote liquid and gas phases respectively. σ denotes the interfacial tension between liquid and gas phases and was set equal to 0.072 N m^{-1}.

According to this relationship, the bubble diameter changes as a function of superficial gas velocity which, in turn, is variable with respect to space. However, bubble diameter does not appear to have a significant effect on the hydrodynamic model, as reported in a previous investigation that the velocity is practically independent of bubble diameter in the 3 to 8 mm range (Van Baten et al. 2003). Inspection of equation 1 furthermore reveals that bubble diameter is insensitive to changes in superficial gas velocity. Increasing and decreasing the superficial gas velocity by a factor of ten leads to a change in bubble diameter of -26% and +32% respectively. Given the small variations in bubble diameter with respect to superficial gas velocity and the small effect of variable bubble diameter on computational output, we consider it acceptable to use one bubble diameter over the entire model domain for a given gas inflow rate (Q_{air}). For a given inflow rate, the bubble diameter used over the flow domain was determined by substituting a single point estimate for the order of the superficial gas velocity into equation 1. This estimated superficial gas velocity was found by dividing gas inflow rate by the area of the riser section, producing values that agree well with experimental data (Behzadian 2010). Table 1.2 sets out the bubble diameters used under different air inflow rates.

Effects of buoyancy between the gas and liquid phases were included, assuming an acceleration due to gravity equal to 9.81 m s^{-2} in the negative z direction and a reference density of 997 kg m^{-3}.

A reference pressure of 0 was used as the pressure datum, so all pressures were absolute.

1.4 Computational model

The Navier-Stokes equations describe exactly the motion of fluids on scales that satisfy the assumption. These equations of conservation of mass and momentum are reproduced respectively in equations 2 and 3.

$$\frac{\partial \rho}{\partial t} + \frac{\partial (\rho u_j)}{\partial x_j} = 0 \quad , \quad j = 1, 2, 3 \tag{2}$$

$$\rho\left(\frac{\partial u_i}{\partial t}+u_j\frac{\partial u_i}{\partial x_j}\right)=-\frac{\partial p}{\partial x_i}+\frac{\partial}{\partial x_j}\left(\nu\frac{\partial u_i}{\partial x_j}\right)-\rho\tau_{ij}+F_i\pm\left(\mathbf{F}_{ME}\right)_i,\ i=1,2,3 \tag{3}$$

where t = time; u_j = j'th component of the Reynolds-averaged velocity vector $\mathbf{u}=<u_1,u_1,u_3>$; x_j = coordinates; p = Reynolds-averaged pressure; τ_{ij} = specific Reynolds shear stress; F_i = the sum of body forces other than momentum exchange in the x_i-direction; $(\mathbf{F}_{ME})_i$ = i'th component of the sum of momentum exchange body forces. Note that the above equations describe the motions of liquid and gas phases separately.They are related through the momentum exchange term, which is equal and opposite between liquid and gas.

The nonlinearity of these equations makes them unsolvable for almost all scenarios of practical interest. Numerical solution is possible by way of so-called Direct Numerical Simulation (DNS). However, the computational mesh in DNS models must cover the entire range of length scales, from the smallest energy-dissipating eddies on the Kolmogorov scale to the largest energy-carrying eddies on the integral scale, because it directly solves all motion in the solution domain. Furthermore, it is not possible to use symmetry boundary conditions to facilitate computation because instantaneous flow is not symmetrical in the turbulent regime (Baumert et al. 2005). Therefore, the computational effort required for DNS increases rapidly with model complexity and is not usually a feasible basis for CFD research. Indeed, alternative models developed throughout the 20th century are vastly less computationally demanding and have been demonstrated to provide good approximations of fluid motion under a wide range of contexts.

The models used in the present research are based on the Reynolds averaged Navier-Stokes equations (RANS) which resolve velocity vectors from equation 3 into average $(\bar{u}_i)$ and instantaneous (u') components:

$$\mathbf{u}=\bar{\mathbf{u}}_\mathbf{i}+\mathbf{u}' \tag{4}$$

The ensuing algebra leaves the Navier-Stokes unchanged except for the substitution of $\bar{u}$ for u and the addition of an apparent stress term to equation 3:

$$\boldsymbol{\tau}_\mathbf{ij}=\rho\overline{\mathbf{u}'_\mathbf{i}\mathbf{u}'_\mathbf{j}} \tag{5}$$

This approach transforms the Navier-Stokes equations into describing motion on the scale of the average velocities only, with the exception of the the apparent stress term described in equation 5. This effectively introduces another unknown into the equations and leads to the well known 'closure problem' of turbulence, which many different models of varying degrees of complexity have been developed to resolve. The family of RANS models used in the present research assume that turbulent eddies behave like molecules, and that their collisions and exchanges of momentum can be described in terms of a so-called eddy viscosity (μ_t) (Ranade 2002). These linear eddy viscosity models define the apparent stress tensor according to equation 6.

$$-\rho\overline{\mathbf{u_i'u_j'}} = \mu_t\left(\frac{\partial\overline{\mathbf{u_i}}}{\partial x_j}+\frac{\partial\overline{\mathbf{u_j}}}{\partial x_i}-\frac{2}{3}\frac{\partial\overline{\mathbf{u_k}}}{\partial x_k}\delta_{ij}\right)-\frac{2}{3}\rho k\delta_{ij} \tag{6}$$

Where $\delta_{ij}=1$ if $i=j$, and $\delta_{ij}=0$ if $i\neq j$, and where k is the turbulent kinetic energy per unit mass. The turbulence models used for the air and liquid phases differ however in how they approximate eddy viscosity.

The well-known k-ε turbulence model was used for the liquid phase. This model accounts for history effects of turbulence (e.g. convection and diffusion of turbulent energy) by describing the eddy viscosity of the liquid phase ($\mu_{t,L}$) in terms of turbulent kinetic energy per unit mass (k) and turbulence dissipation rate (ε) according to equations 7 to 9 (ANSYS 2009).

$$\mu_{t,L}=C_\mu\rho\frac{k^2}{\varepsilon} \tag{7}$$

$$\frac{\partial(\rho k)}{\partial t}+\nabla\cdot(\rho k\mathbf{u})=\nabla\cdot\left[\left(\mu_L+\frac{\mu_{t,L}}{\sigma_k}\right)\nabla k\right]+P_k+P_{kb}-\rho\varepsilon \tag{8}$$

$$\frac{\partial(\rho\varepsilon)}{\partial t}+\nabla\cdot(\rho\varepsilon\mathbf{u})=\nabla\cdot\left[\left(\mu_L+\frac{\mu_{t,L}}{\sigma_\varepsilon}\right)\nabla\varepsilon\right]+\frac{\varepsilon}{k}\left(C_{\varepsilon1}\left(P_k+P_{\varepsilon b}\right)-C_{\varepsilon2}\rho\varepsilon\right) \tag{9}$$

Where $\mu_{t,L}$ is the molecular viscosity of the liquid phase and C_μ, σ_k, σ_ε, $C_{\varepsilon1}$ and $C_{\varepsilon2}$ are constants. P_k represents the production of turbulence by viscous forces and is modelled as in equation 10. P_{kb} and $P_{\varepsilon b}$ represent the production and dissipation of turbulence by buoyancy, respectively, and are modelled as in equation 11 and equation 12.

$$P_k=\mu_t\nabla U\cdot\left(\nabla U+\nabla U^T\right)-\frac{2}{3}\nabla\cdot U\left(3\mu_t\nabla\cdot U+\rho k\right) \tag{10}$$

$$P_{kb}=-\frac{\mu_t}{\rho\sigma_\rho}g\cdot\nabla\rho \tag{11}$$

$$P_{\varepsilon b}=max\left(0,P_{kb}\right) \tag{12}$$

Because the liquid phase is considered to be isothermal and incompressible, having everywhere a density gradient of 0, there is no buoyancy and hence no buoyancy turbulence.

Whereas the foregoing equations are solved numerically to approximate the stress tensor for the liquid phase, a simpler model was used for the dispersed bubbles: a dispersed-phase zero-equation model was applied to the gas phase, which defines gas eddy viscosity ($\mu_{t,g}$) in terms of liquid eddy viscosity as in equation 13.

$$\mu_{t,g} = \frac{\mu_{t,L}}{\sigma_{Pr}} \tag{13}$$

Where σ_{Pr} is a turbulent Prandtl number.

If the particle relaxation time of the dispersed gas phase is much smaller than the turbulent dissipation time scale, T_D, it is permissible to set the kinematic eddy viscosity of the gas phase equal to that of the liquid phase, i.e. to assume a turbulent Prandtl number equal to 1 (ANSYS 2009). In the case where τ approaches T_D it is preferable to assume a turbulent Prandtl number greater than 1, hence decreasing the estimate of the kinematic eddy viscosity of the gas phase relative to that of the liquid phase.

Equations 14and 15 describe the particle relaxation time (Ranade 2002) and turbulent dissipation time scales, respectively, where the latter is approximated by the ratio between the diameter of the aerobic riser zone (D_a) and the rising gas velocity. Rising gas velocity was in turn calculated by dividing superficial gas velocity by the gas hold-up (ε_g), where the order of the superficial gas velocity is estimated as for the bubble diameter calculation based on equation 1. This provides an estimate for the timescales of the largest and most energy-carrying turbulent eddies, namely, those with a length scale on the order of the diameter of the riser column and a velocity scale on the order of the speed of the rising gas.

$$\tau = \frac{d_b^2 \rho_g}{18 \mu_L} \tag{14}$$

$$T_D \sim \frac{1}{|\mathbf{u}|} \sim \frac{D_C}{|\mathbf{u_{g,r}}|} = \frac{D_a \in_g}{|\mathbf{u_{g,s}}|} \tag{15}$$

Several equations for calculating gas holdup as a function of superficial gas velocity are in wide use for different geometries and conditions (Tabib et al. 2008). Hughmark provides one such equation applicable to the dimensions of the riser column, reproduced as equation 16 (Ishii and Hibiki 2011).

$$\epsilon_g = \left[2 + \left(\frac{0.35}{|\mathbf{u_{g,s}}|} \right) \left(\frac{\rho_L \sigma}{72} \right)^{\frac{1}{3}} \right]^{-1} \quad (16)$$

The particle relaxation time is a function of the slip velocity between air and liquid phases, which varies in space in a way that is largely unknown *a priori*. In order to produce a conservative (high) estimate for τ, we consider a minimum slip velocity. This is taken to be equal to the difference between the maximum liquid velocity and minimum gas velocity, under the assumption that the gas phase has everywhere a higher velocity than the liquid phase, on the basis that the liquid phase is agitated by inflowing air. The maximum liquid velocity considered is equal to the point estimate for gas rising velocity, higher than the velocity actually acquired by the liquid phase via the mechanism of momentum exchange, but a satisfactory approximation for this purpose. The minimum gas velocity considered is equal to the highest velocity attained under acceleration due to buoyancy with an initial velocity equal to the gas rising velocity at the sparger. The significant effects of momentum exchange between the gas and liquid phases, which significantly slows the gas, is neglected in order to produce a conservative (high) estimate for maximum gas velocity.

Table 1.2 presents the results from the calculations and finds the particle relaxation time to be roughly an order of magnitude smaller than the turbulent dissipation timescales. We are therefore justified in setting the turbulent Prandtl number equal to 1 in the dispersed-phase zero-equation turbulence model used for the gas phase. This analysis is only relevant to the riser column, because the gas holdup and hence the effects of momentum transfer and buoyancy-induced turbulence are very small in the other regions of the reactor.

The rise velocity of an air bubble within a continuous liquid phase is determined by its exchange of momentum with the liquid phase, as stated in equation 17.

$$m_b \frac{d\mathbf{u_{r,g}}}{dt} = \mathbf{F_{ME}} = \mathbf{F_D} + \mathbf{F_L} + \mathbf{F_{TD}} + \mathbf{F_{WL}} + \mathbf{F_{VM}} \quad (17)$$

Where m_b is the mass of an individual bubble under analysis and $\mathbf{F_{ME}}$ is the sum of momentum exchange forces, including buoyancy. $\mathbf{F_D}$, $\mathbf{F_L}$, $\mathbf{F_{TD}}$, $\mathbf{F_{WL}}$ and $\mathbf{F_{VM}}$ represent drag, lift, turbulent dispersion, wall lubrication and virtual mass components of momentum exchange, respectively. As indicated in equation 3, this term is equal and opposite in the liquid and gas phases. Virtual mass force was however neglected; other bubble column modellers have concluded that the marginal benefits of accounting for the virtual mass force are outweighed by convergence problems and significantly longer computation times (Kamali et al. 2007). Similarly, we have omitted the wall lubrication component of momentum exchange, as the improved precision in the near-wall region is likely to be insignificant given the error introduced by considering a free-slip wall boundary condition and therefore not worth the added computational effort.

Drag force between a particle and the continuous fluid surrounding it is defined by equation 18.

$$\mathbf{F_D} = \frac{1}{2} C_D \rho_L A_B \left(\mathbf{u_{r,g}} - \mathbf{u_L} \right) \tag{18}$$

Where A_B is the effective bubble diameter and C_D is the drag coefficient. Instead of prescribing one value of C_D for all grid points and timesteps in the simulation, the Grace model was applied (Yeoh & Tu 2010). The drag coefficient was thus calculated individually at every grid point and timestep according to equation 19.

$$C_D = \frac{4}{3} \frac{g d_b}{\left| \mathbf{u_T} \right|^2} \frac{\rho_L - \rho_g}{\rho_L} \tag{19}$$

Where $\mathbf{u_T}$ is terminal velocity as described in equations 20 to 22.

$$\left| \mathbf{u_T} \right| = \frac{\mu_L}{\rho_L d_b} \left[\frac{\mu_L^4 g(\rho_L - \rho_g)}{\rho_L^2 \sigma^3} \right]^{-0.149} \left(J - 0.857 \right) \tag{20}$$

$$J = \begin{cases} 0.94 H^{0.751} & 2 < H \le 59.3 \\ 3.42 H^{0.441} & H > 59.3 \end{cases} \tag{21}$$

$$H = \frac{4}{3} \left[\frac{g(\rho_L - \rho_g) d_b^2}{\sigma} \right] \left[\frac{\mu_L^4 g(\rho_L - \rho_g)}{\rho_L^2 \sigma^3} \right]^{-0.149} \left[\frac{\mu_L}{\mu_{ref}} \right]^{-0.14} \tag{22}$$

The reference viscosity, μ_{ref}, in equation 22 is that of water at 25°C. Because the liquid phase is modelled as water at 25°C, the quotient μ_L/μ_{ref} equals 1.

CFX supports different models for lift generated by rotational movement of one fluid around the other, in this case, rotational movement of water around elements of air. The standard equation for lift is reproduced in equation 23.

$$\mathbf{F_L} = \rho_L C_L \varepsilon_g \left(\mathbf{u_{g,r}} - \mathbf{u_L} \right) \times \left(\nabla \times \mathbf{u_L} \right) \tag{23}$$

Where C_L is the lift coefficient. The model proposed by Legendre and Magnaudet for spherical bubbles in viscous linear shear flow (Legendre & Magnaudet 1998) was used to calculate C_L at every timestep. The authors produce lift coefficients for different strain rates in the range range $0.1 \le \mathrm{Re} \le 5$ and demonstrate independence of strain rate for higher Reynolds numbers as well as a general flattening out of the lift coefficient through Reynolds numbers on the order of 1000.

The empirical correlation between Reynolds number and lift coefficient for the range Re > 5 is provided in equation 24.

$$C_L = \frac{1}{2}\frac{1+16Re^{-1}}{1+29Re^{-1}} \qquad (24)$$

The standard Lopez de Bertodano lift model was used, as reproduced in equation 25 (Lopez de Bertodano 1998):

$$\mathbf{F}_D = C_T \rho_L k \nabla (1-\varepsilon_g) \qquad (25)$$

Where C_{TD} is the turbulent dispersion coefficient. The variable k describes turbulent kinetic energy of the liquid phase for both liquid and gas phase momentum exchange equations. Equation 26 describes the calculation of C_{TD}. Table 1.2 presents the values calculated for C_{TD} the present model.

$$C_{TD} = \frac{C_\mu^{1/4}}{\frac{\tau}{T_D}\left(1+\frac{\tau}{T_D}\right)} \qquad (26)$$

Where C_μ is the same constant as in equation 7. The k-ε model sets this value to 0.09.

1.5 Boundary conditions

Boundary conditions were programmed so as to approximate the behaviour of the laboratory-scale reactor.

Conditions at the air sparger were approximated by modelling uniform airflow through the surface of the sparger at the base of the aerobic zone. Air inflow speeds were calculated to correspond to bulk flows of 10, 30, 50 and 70 litres per minute across the disc area of 0.02 m² in the case of the cylindrical sparger and 0.01 m² in the case of the torus sparger (air inlet defined as the surface a 45° wedge cut into the torus). The liquid phase inflow fraction was set to 0. We note that this does not have the effect of setting the boundary condition to 0% water at the sparger; after the simulation, it was found that within one grid space, the air fraction at the height of the sparger is close to the approximate values listed in Table 1.1 and the experimental values reported in by Behzadian (2010).

CFX allows for the values of *k* and *ε* to be set manually at the inlet, or to automatically scale the intensity of turbulence based on the ratio between the magnitudes of the fluctuating and average parts of the Reynolds averaged Navier-Stokes (RANS) equations. This ratio was set to the default of 5%.

The entire surface of the reactor, with the exception of a small wedge used for wastewater inlet, was considered to be open to air at 1 atm allowing for exit of the gas phase, but not the liquid phase.

A 5-cm-radius wedge in the middle of the reactor was considered as the liquid-phase inlet area. The inflow was varied between 0 to 720 L day^{-1} to study its effect on the hydrodynamic characteristics of the reactor.

At the top of the clarifier zone, defined as the liquid outlet boundary, a pressure of 1 atm and normal flow direction was specified.

A free-slip wall was considered for the liquid and gas phases, based on the coarseness of the mesh size relative to the dimensionless wall distance.

Two planes of symmetry allowed a 90-degree wedge of the reactor to serve as a model for the entire cylindrical shape.

1.6 Model runs and results

The model was run a total of 25 times with varying operating conditions and geometry to study the resulting differences in hydrodynamic characteristics of the reactor. The runs were roughly grouped in order to examine the effects of modifying the value of a single input parameter while holding the others constant. Runs 1 to 8 modelled the air sparger as a solid cylinder, and Runs 9 to 25 modelled the sparger as a torus. Runs 1 and 3 to 5 (cylinder sparger) and 9 to 12 (torus sparger) varied the air inflow rate while holding the wastewater flow rate constant. Runs 6 to 8 (cylinder sparger) and 13 to 18 (torus sparger) varied the wastewater flow rate while holding the air inflow rate constant at 30 L min^{-1}. Runs 19 to 25 varied the geometry in other ways while holding both air and wastewater flow rates constant. Runs 8 (cylinder sparger) and 18 (torus sparger) used the geometry and operating conditions from the reactor set-up, including a wastewater flow rate of 720 L day^{-1}. The values of all parameters for each run are indicated in Table 1.1, and the select hydrodynamic characteristics in Table 1.3.

The output of the runs most closely approximating the experimental set-up was compared to experimental results. Liquid circulation time around the aerobic and microaerophillic zones was calculated from reactor output data for these two runs and compared with the circulation time calculated experimentally. The experimental liquid circulation time at air inflow rate of 30 L min^{-1} was 33 s, and dependence on wastewater flow rate was not studied in the laboratory (Behzadian 2010). A circulation time of 90.2 s was calculated from the output of Run 8 (cylinder sparger) and 48.9 s from Run 18 (torus sparger), both of which modelled wastewater flow rates of 720 L day^{-1}. Run 6 (cylinder sparger) and Run 13 (torus sparger) yielded circulation times of 31.3 s and 25.10 s, respectively. The effects on the circulation time of varying the wastewater throughput rate were further studied by conducting Runs 6 and 7 for the cylinder sparger and Runs 13 to 17 for the torus sparger. Although runs conducted with no wastewater flow yield circulation times agreeing better with experimental results, we see no correlation between wastewater flow rate and circulation time.

Experimental results showed circulation time to be a parameter very sensitive to geometry and air inflow rates. For example, reducing the diameter of the port between the aerobic and microaerophillic zones by 50% leads to an increase in liquid circulation time of between 174% and 205% for the range of air inflow rates studied. The variations in calculated circulation time among runs with a variable wastewater flow rate are all less than ± 50% around the mean values of 61.3 s (cylinder sparger) and 41.7 s (torus sparger). Because the variations in circulation time calculated are small compared to the variations caused by minor changes in reactor geometry, and because they show no trend with respect to the wastewater flow rate, we consider the variations we observe to be artefacts of the computational model. We further observe that the circulation time calculated from the model compares very well with the experimental results, particularly where the sparger is modelled as a torus.

Further comparisons were made between the average vertical velocities in the aerobic zone calculated from the model and values suggested by Chisti et al. (1988) and Bello et al. (Bello et al. 1984). The average velocities for Runs 6 and 13, which had no wastewater flow, air inflow of 30 L min^{-1} and used cylinder and torus sparger geometry respectively, are both within 5% of the value of 0.15 m/s suggested by Chisti et al. (Chisti et al. 1988). The other runs agree reasonably well in the same way, and all but three show average vertical velocities within 50% of the values suggested by Chisti et al. for the relevant air inflow rates (Chisti et al. 1988). The velocities suggested by Bello et al. are roughly an order of magnitude higher than the model output (Bello et al. 1984). We observe no correlation between average vertical velocity in the aerobic zone and prescribed wastewater inflow rate, and the differences between Runs 6 to 8 (cylinder sparger) and 13 to 18 (torus sparger) are likely artefacts of the computational model.

The clarifier zone is considered to have no dispersed gas fraction, an assumption easily validated by checking the clarifier gas holdup in the model output. Therefore, the theoretical mean vertical velocity in the clarifier zone is equal to the liquid throughput, equal to the wastewater throughput, divided by its cross sectional area. Even at maximum flow rate, the theoretical average velocity is less than 1E-04 m s^{-1}, too small to be adequately resolved by the model used. The mean vertical velocities computed in the clarifier zone are therefore not reliable measures of model performance or accuracy.

Indeed, the output of several runs indicates a mean vertical velocity in the clarifier that is slightly negative, i.e. downward, despite the mean upward direction being required by the principle of continuity. The velocity residence times calculated for the clarifier are not included in Table 1.3.

Circulation time and mean velocities are important for judging the accuracy of the model we have developed as well as for evaluating microbiological treatment and wastewater characteristic requirements implied by other designs the model can be adapted to simulate. Similarly, hydrodynamic characteristics such as turbulence, circulation pattern and flow direction have implications for model performance and loading and treatment requirements under different operating conditions. This study examines these other hydrodynamic characteristics only qualitatively.

In all runs, we see upward flow in the core of the aerobic zone with downward flow near the wall. Even at maximum wastewater flow rates from the top of the aerobic zone, the flow direction below the outlet to the microaerophillic zone is not affected. Flow in the microaerophillic zone is downward with some circulation at and above the height of the outlet from the aerobic zone. At the base of the microaerophillic zone, the flow separates. Part of the flow field from the microaerophillic zone is continuous with the flow field leading upwards into the aerobic zone past the sparger, and part disperses into large eddies in the conical anoxic zone. These eddies in turn appear to have a large impact on the flow pattern in the clarifier zone.

The mean vertical velocity in the clarifier zone was calculated to be many orders of magnitude less than in the aerobic and microaerophillic zones. The flow is therefore easily perturbed by upstream activity, around the edges of the anoxic cone. Flow in the anoxic zone just below the clarifier is roughly radial and results from the broader pattern of eddies in that area. In Runs 1 to 20, 23 and 25, the 7-cm-wide clarifier is fully open to the perpendicular flow below. In these runs, the flow pattern in the clarifier demonstrates pronounced circulation. The geometry in Runs 21 and 22 is modified variously so as to partially shield the entrance to the clarifier from the shearing horizontal eddies in the anoxic zone. The resulting flow pattern is very steady, uniform upward flow. Ensuring this flow pattern in the clarifier is important from a treatment perspective in order to allow for easy settling of solids.

Runs 13 to 18 varied the wastewater inflow rate in order to determine whether higher flows had a smoothing effect on the velocity field in the clarifier. Even at very high flow rates, however, the flow pattern in the clarifier exhibited significant circulation. Therefore, the only effective mechanism for inducing smooth upward flow in the clarifier was modification of skirt geometry so as to shield the flow within from the circulation in the cone below.

Figures 1.3 to 1.9 were generated with R in order to compare Runs 10 and 22, which differ only with respect to the geometry of the skirts around the base of the clarifier. Figures 1.3 and 1.4 plot z-direction velocities versus radial distance at three heights: 0.12, 0.43 and 0.73 m above the sparger. Figure 1.3 corresponds to Run 10 (without clarifier isolation) and Figure 1.4 corresponds to Run 22 (with clarifier isolation). Because the model results for velocity values in the clarifier are too small to be reliably quantified, they are omitted from the figures. The velocity distributions in the aerobic (blue) and microaerophilic (red) zones are very similar.

Figures 1.5 and 1.6 are set up in the same way as Figures 1.3 and 1.4, but the velocity data correspond to normalized vectors in the x- and y-directions. The velocity magnitudes in Figures 1.5 and 1.6 are therefore a two-dimensional projection of the three-dimensional velocity vector onto the x-y axis. A value of 0 indicates that the three-dimensional velocity vector points entirely in the z-direction (unless it too is 0). Because the principle of continuity dictates that the average direction of fluid motion is positive-z in the aerobic zone and negative-z in the microaerophilic zone, the magnitude (and range of magnitudes) of the x-y projection of the velocity vector indicates the circulation occurring at the point studied. Again, we see substantial similarity between Runs 10 and 22. We also see that the greatest circulation in the aerobic zone occurs nearest to the sparger, and that the greatest circulation in the microaerophilic zone occurs nearest to the spout connecting the two zones. Circulation in the microaerophilic zone dissipates rapidly as fluid moves down. The spread of velocities observed at the wall between the microaerophilic zone and the clarifier is likely to be an artefact of the computational model.

Finally, Figures 1.7a, 1.7b and 1.7c compare the probability distributions of the magnitude of the x-y projection of the velocity vector at different heights between Runs 10 and 22. In this case, the figures are separated according to height above the sparger (0.12, 0.43 and 0.73 m) with Runs 10 and 22 compared on each. The y-axis represents the probability density of each 'bin' of velocity ranges. Although the precision of the computational model was not adequate to reliably quantify the magnitudes of the velocity vectors in the clarifier, we observed that the overall flow pattern conformed to predictions about the impact of changing geometry and wastewater flow rate. To roughly represent the circulation observed in Run 10 (without clarifier isolation) and Run 22 (with clarifier isolation), we plot the distribution of the x-y projection of the velocity vector. Indeed, we see that the bulk of the distribution is significantly lower in Run 22 than in Run 10, particularly at a height of 0.43 m above the sparger (Figure 1.7b). The reduction of circulation is less significant at the bottom (Figure 1.7a) and the top (Figure 1.7c) of the clarifier, although this might be an artefact of the computational model.

Figures 1.8 and 1.9 show the flow pattern in the unprotected clarifier modelled in Run 10. Figures 1.10 and 1.11 show the flow pattern in the shielded clarifier modelled in Run 22. They are included in the supplementary information section.

1.7 Conclusion

This research has applied the methods of computational fluid dynamics to study the principle hydrodynamic characteristics of a new type of wastewater treatment system. The novelty of the technology studied lies primarily in the integration of an external clarifier zone, isolated from the air-driven flow in the aerobic and microaerophillic zones. This feature has led us to explore possible effects of wastewater flow rates, a dimension not studied in other works of this category. Model output was compared with experimental results to confirm its validity and then used to explore the effects of varying operating conditions and geometries on circulation time, mean vertical velocities and the qualitative characteristics of flow patterns.

The model we developed produces results that compare well with experimental data over a range of operating conditions. The model suggests that the hydrodynamic characteristics are essentially independent of wastewater flow rate used, even in the clarifier zone where the wastewater throughput is responsible for the mean flow. Perhaps most importantly, the geometry of the reactor around the base of the clarifier has a direct effect on the smoothness of flow towards the exit and must be chosen so as to be conducive to the settling of solids.

Hydrodynamic characteristics such as circulation and flow pattern which in this research have been described only qualitatively, and characteristics such as turbulence, which are not examined in this research, can be quantified in future study of the reactor configurations of interest. Primarily, the reactor configurations that suggest smooth rising in the clarifier should be explored in greater detail to model more quantitatively the hydrodynamic characteristics that might have an impact on the treatment potential of this technology: this research found that adjusting reactor geometry is far more likely to have an impact on flow pattern in the clarifier than modifying operating conditions.

Although the body of literature on CFD modelling of air-lift bioreactors is relatively mature, this research is unique in its study of the hydrodynamics around the integrated clarifier zone. The primary contribution of this research is to provide guidance on the configuration of the skirt isolating the clarifier. This research also demonstrates that the conclusions drawn on the configuration of the clarifier geometry are independent of operating characteristics such as air flow and wastewater throughput rates. Therefore, these conclusions are very easily portable to other bioreactors that feature an integrated clarifier zone.

It would be possible to expand the model to consider a dispersed solid phase. This would be useful for modelling sedimentation and the behaviour of the solid phase itself. The momentum transfer to the solid phase would also have a small slowing effect on the circulating liquid phase. Results from the present study imply that inclusion of a solid phase would not have any bearing on the conclusions with respect to the importance of skirt geometry below the clarifier zone. This study already examined the effects of a variable wastewater throughput rate on the hydrodynamics of the clarifier zone and found them to be negligible compared to the effects of geometry. Because the dispersed solid phase would have an even smaller impact on the momentum exchange in and around the clarifier zone than the wastewater throughput rate, we can conclude that a dispersed solid phase would not have a measurable effect on the hydrodynamics of that zone.

CHAPTER 2

A probabilistic decision-making framework for the management of sewage plumes in a tidal environment[2]

The complex and unsteady hydrodynamics of coastal waters greatly complicates water quality assessment and prediction and hence impact valuation and decision-making. A rational analysis requires an understanding of underlying hydrodynamics, principles of water quality in unsteady environments, impact valuation methods and decision analysis techniques. Two-dimensional numerical models of sewage plumes in tidal environments illustrate the high sensitivity of water quality to input parameters such as density stratification and tidal condition that vary continuously with time. This chapter demonstrates the applicability of probabilistic methods in translating discrete snapshots of water quality under time-variable conditions into a description of water quality that can be used to evaluate the costs of ecological and economic impacts. In order to allow for comparison of management alternatives in terms of the net costs of each (infrastructure investment and the associated ecological and economic impacts), the quality/impact valuation model is nested in a decision analysis framework. The case study fleshed out starts with a numerical water quality model of Burrard Inlet near Vancouver, Canada. This model is a good example of the state of the art in environmental fluid mechanics and combines and describes water quality under multiple discrete combinations of input conditions.

2.1 Introduction

The complex physical processes at play in coastal waters make them among the most economically and socially valuable habitats on Earth, a fact long since recognized by humans; over 60% of the world's population is settled on lands adjoining estuaries (Wolanski 2007). The fruitful physical complexities and high population density of estuarine environments have two corollaries especially relevant for engineers and policy-makers: these waters bear the brunt of anthropogenic environmental impacts, and, as importantly, these impacts are extremely difficult to model and therefore manage.

This chapter constructs a decision model for the management of sewage discharges to tidal waters that integrates computational fluid dynamics (CFD) model output with probabilistic and impact valuation analysis. This framework has the potential to substantially improve water quality management and decision-making strategies by making better use of CFD models.

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[2] This research was presented at the Society for Risk Analysis 2012 annual conference in Charleston, South Carolina. It won the Ecological Risk Assessment Student Merit Award.

The example presented here is based on the case study of Burrard Inlet, near Vancouver, Canada, where sewage behaviour was studied in a CFD model by Li & Hodgins (2010). Their research models the sewage plume in the receiving waters under a number of discrete combinations of input conditions. The example fleshed out in this work is an infrastructure decision, where managers must weigh the environmental impacts of sewage discharges, the frequency of impacts, the cost of their occurrence and the cost of their abatement for several alternative courses of action. A conceptual overview of the model is included in Figure 2.1. Our task is therefore to demonstrate how probabilistic risk assessment and impact valuation methods can be applied to 1) resolve the CFD model of discrete conditions into an average index of water quality and 2) use the water quality measure obtained for different CFD simulations to inform environmental management decision-making. Specifically, this case study supposes two hypothetical policy alternatives: 1) do nothing and 2) complete risk abatement. In the do-nothing alternative, the risk model is applied to water quality data that reflects the status quo. Cost of impacts is a function of a valuation model and cost of investment is zero. Likewise, the cost of impacts in the complete risk abatement alternative is zero and cost of investment is some dollar value. In fact, we do not define this dollar value *a priori*, but rather consider that it must be at most equal to the cost of the risk from the do-nothing alternative; the risk cost from the do-nothing alternative is logically equal to the highest cost warranted for complete risk abatement.

2.2 Background

Discharge to receiving waters of untreated or partially treated sewage is a relatively common feature of the Canadian landscape of wastewater management, even in densely populated parts of the country (Natural Resources Canada 2004). Water quality problems appear when the loading exceeds the assimilative capacity of the receiving waters and can manifest themselves as surfacing, settling or washing ashore of parts of a sewage plume, adversely affecting the recreational or economic value of aquatic or coastal environments. The ability to model and therefore predict such water quality problems allows for a timely and well-informed engineering or policy response and is therefore of great interest to actors in environment and public infrastructure.

Hydrodynamic models have recently been developed to plot sewage plumes in tidal environments and illustrate the high sensitivity of the plume shape and dilution to time-variable input parameters. Wastewater loading, tidal condition and seasonal stratification vary continuously with time and different combinations of assumptions produce dramatically different results. The Li & Hodgins model of Burrard inlet, which we use to develop a case study of our decision-making framework, provides 12 two-dimensional plume profiles that represent each combination of representative assumptions for wastewater loading (present-day dry weather, present-day wet weather), tidal condition (ebb, flood, slack) and seasonal stratification (summer, winter). While this hydrodynamic modelling approach provides valuable insight into the spatial distribution of sewage plumes and its dependence on time-variable input conditions, a complete water quality assessment can only be made with an understanding of the frequency/probability of each of the 12 ‘representative’ plumes and of the interaction between the different plumes and the environment to produce quantifiable impacts.

The advanced modelling effort required to characterize water quality in tidal waters and the probabilistic nature of water quality model output present significant barriers to rational public policy decisions in matters of wastewater treatment and water management planning. Decision-making must be guided by a synthetic water-quality/economic-valuation model that identifies potential impacts, calculates the probability of their occurrence and quantifies the cost of occurrence as compared to the costs of various policy responses.

Beyond the essentially deterministic input parameters studied for example by Li & Hodgins lie additional challenges posed by stochastic phenomena such as wind shear, non-tidal transport waves, turbulent mixing and dispersion. This research organizes the deterministic Li & Hodgins CFD model into a probabilistic framework and does not directly treat stochastic parameters.

The study by Li & Hodgins integrates near- and far- field CFD models, with near-field being used in the vicinity of the sewage outfall and far-field being used farther upstream and downstream. The near-field model is based on the line-buoyant plume model UM, developed by US EPA. The far-field model uses large eddy simulation (LES). The two models are integrated so as to eliminate discontinuities at the boundary. Model output results were found to be particularly sensitive to density stratification, sewage discharge rate (a function of weather) and the directionality of ambient flow. Model results were therefore evaluated over each combination of representative values for each significant parameter.

2.3 Risk model for water quality

Current CFD models express water quality and sewage behaviour under discrete combinations of inputs that describe receiving water conditions and sewage loading. Our first task is to understand how the discrete, representative model outputs combine to describe water quality on average. We apply concepts from probabilistic risk assessment to organize the discrete snapshots of water quality according to their probability/frequency. Once this is complete, an impact valuation model can be applied in order to produce information usable for decision-making.

2.3.1 Event tree for as-is scenario

We illustrate the development of an event tree by constructing one for the Li & Hodgins model of Burrard Inlet. The model was run for 12 discrete cases: one for each combination of assumptions for water stratification (summer or winter), tidal motion (ebb, flood or slack) and sewage loading rate (1 or 3 m^3/s for dry and wet weather respectively). These profiles output two-dimensional dilution curves and are designed to represent plume dilutions as a function of geometry under the spectrum of physical conditions in the tidal water. They ultimately aim to evaluate under what conditions sewage plumes exhibit undesired behaviour, the most relevant characteristics being surfacing, settling and far longitudinal transport of undiluted sewage.

Considering the variables to be independent, we can calculate marginal probabilities from average weather and tidal data (Environment Canada 2010; Fisheries & Oceans Canada 2011).

Variable 1: Seasonal stratification

$p(season = summer) = 0.5$

$p(season = winter) = 0.5$

Variable 2: Tidal variation

$p(tides = ebb) = 0.67$

$p(tides = slack) = 0.04$

$p(tides = flood) = 0.29$

Variable 3: Dry weather/wet weather flow rate

$p(weather = dry) = 0.55$

$p(weather = wet) = 0.45$

Figure 2.2 organizes the 12 cases from Li & Hodgins according to variables in their model input and the real probabilities we assign each of them. Note that the summer/winter and ebb/slack/flood variables have a direct physical bearing on the water quality model attributable to receiving water stratification and tidal motion, respectively. Dry and wet weather represent differences in sewage loading rate: 1 m^3/s in dry weather and 3 m^3/s in wet weather. Sewage loading rate is the underlying parameter that is used as input in the water quality model. The probability of each loading rate is therefore decided by historical weather data.

2.3.2 Impact valuation model

Surfacing, settling, washing ashore and longitudinal transport of undiluted sewage have diverse impacts on the ecological, economic and aesthetic value of receiving waters. Impact quantification is highly site-specific, and conducting this kind of analysis for our case study of Burrard Inlet is beyond our scope. A review of the literature did not uncover any methods for translating time-variable water quality into an average index that could be useable for water quality forecasting, impact valuation and decision-making. Rather, site-specific studies have tended to be undertaken in order to understand impacts retrospectively. This may be because there has so far been no framework for improved decision-making on the basis of more sophisticated valuation models. This represents a significant barrier to developing a decision model for tidal waters. The present research may however provide a basis for further research in that direction.

To move forward in our framework, we consider a simplified, temporary impact valuation method based on several simplifying assumptions:

1. Environmental and economic receptors are adversely affected by a sewage plume that surfaces, settles or is transported far downstream. Sewage that rapidly mixes, dilutes and washes to sea has no impact.
2. Environmental and economic impacts are measured by changes in recreational and fishing value of the water and changes in property values of land adjoining the water.
3. Fishing value of the water is negatively impacted by surfacing, settling and far longitudinal transport of undiluted sewage; tourism and recreational value of the water and property values of adjoining land are negatively impacted only by surfacing of undiluted sewage.
4. Undiluted is considered to mean at least as concentrated as one part in one hundred thousand (10^{-5}). Concentrations equal to or greater than 10^{-5} affect receptors equally, and concentrations smaller than this have no impact.
5. The value of the impact is proportional to its probability/frequency, regardless of the timing. A receptor has zero value when it is impacted according to the foregoing assumptions and full value otherwise.
6. Property values on the land adjoining the water are only partially a function of the water quality. It is assumed that degraded water quality can impact property values to a maximum of 10%.

In addition to allowing us to move forward with constructing a decision-making framework, this simplified impact model demonstrates the questions that a realistic valuation model would need to address in order to implement this tool.

With respect to the environmental and economic valuations addressed in points 1 and 2, this framework depends on the use of tools that translate degraded water quality into economic impacts and also on tools that translate plume behaviour into the specific phenomena that are responsible for degraded water quality. Eutrophication and algal bloom are significant biochemical processes that degrade water quality and are responsible for decreased recreational and economic value. Impacts of these phenomena are relatively well referenced in the context of steady-state water systems (e.g. Anderson et al. 2000; Hoagland et al. 2002). Likewise, models relating nutrient and phosphorus content of receiving waters to resulting eutrophication and proliferation of algae (Chapra 1996). However, in order for the framework established in this chapter to move forward, analogues to these models will need to be established in the context of transient water bodies.

2.3.3 Sample model output

The CFD output by Li & Hodgins was combined with the risk and provisional valuation model described above to flesh out the decision-making framework. The results are tabulated in Table 2.1.

The several binary simplifications described in the preceding section allow for the value of each receptor to be proportionally reduced according to the probability/frequency of receiving water behaviour that impacts it. This therefore assumes knowledge of the maximum potential value of each receptor in the absence of receiving water contamination.

The risk and provisional valuation model calculates an 85% reduction in the value of fishing activities, a 57% reduction in the recreational use value and a 6% reduction in property values of the land nearby. With data on the best-case value of the receptors, these results would facilitate a rational decision as to the maximum risk investment is warranted in risk abatement.

Although the provisional valuation model has made several simplifying assumptions, it has brought us closer to accounting for costs of impacts in a highly transient water body. Indeed, we can see from Table 2.1 that the worst-case results (CFD runs 5, 7, 10-14 and 17) account for only about 57% of expected outcomes, meaning that a large proportion (43%) of an annual period of analysis would present light or no impacts. It is likely worth refining the valuation model in order to take proper account of this impact-free time to avoid overestimating the value of impacts as would do a valuation model based on the worst case scenario.

2.4 Chance and future predictions

So far, we have described the process of assessing and valuing impacts of an existing scenario with known input conditions and no element of chance. Recall that the rightmost leaves on Figure 2.2 associated with the probability of dry and wet weather are the determinants of sewage loading rates (1 and 3 m^3/s, respectively). These sewage loading rates are used as input in the CFD models, and plume profiles are generated. The probabilities associated with these discharge rates are thus just the frequencies of dry and wet weather; the discharge rates are known for each of them.

Because infrastructure decisions made now must consider forecasted conditions, current loading rates are not enough to produce a completely meaningful decision model. Future loading rates, which *do* have an element of chance, are needed. Figure 2.3 illustrates the expansion of the decision model to handle probabilistic predictions of future contaminant plumes, zoomed in on the summer/ebb branch. The probabilities of 'dry weather' and 'wet weather' branches would remain as they are currently, but they would no longer directly determine the wastewater discharge rate used in the CFD model. Instead, predictions 1 to 3 (or more) would be estimates of future discharge rate, as calculated based on population predictions, future infrastructure loading, or other indicators. After running the CFD model under these hypothetical model conditions and applying the valuation model, traditional risk analysis techniques can be used to inform decision-making.

In the previous section, we reviewed the complexities of water quality modelling in estuarine environments. Beyond the wide variation in deterministic parameters that have significant impact on the model output, there are other influences that require probabilistic treatment in the water quality model themselves. What we have seen so far has been probabilistic treatment of divergent deterministic water quality predictions. Additionally, refined water quality models that are themselves probabilistic expressions can be resolved into the decision-making framework if the CFD models themselves are enhanced. The CFD models nested in the decision-model would need to run themselves under different input conditions and identify and store those results that are relevant to decision-making. Using our sample valuation model, the enhanced CFD code should count simulations whose match certain criteria: 1) surfacing, 2) settling and 3) far transport. The probability of different impacts for different assumptions can thus be calculated as a part of the water quality model based on the number of simulations out of the total number that resulted in impacts as we are defining them. This approach can be considered as nesting CFD within a Monte Carlo simulation. Alternatively, we could adapt the CFD model to accommodate stochastic parameters internally. Lin et al. claim that such an internally stochastic CFD model is more than an order of magnitude more computationally intensive than its deterministic equivalent (2007). Because our CFD models have several locations of interest (surface, bottom, etc.), they might be considerably more intensive than this. We would however expect that the alternative of nesting a deterministic CFD model within a Monte Carlo simulation would represent much more than ten times the effort required to model the same deterministic simulation once. Therefore, if the CFD model can be easily adapted to accommodate stochastic parameters for points of interest, it might be more efficient to create internally stochastic simulations.

CHAPTER 3

The role of detection limits in drinking water regulation[3]

Some commentators on environmental science and policy have claimed that advances in analytical chemistry, reflected by an ability to detect contaminants at ever-decreasing concentrations, lead to regulations stricter than justified by available toxicological data. We evaluate this claim in the context of drinking water regulation, with respect to contaminants regulated under the Safe Drinking Water Act (SDWA). We examine the relationships between historical and present maximum contaminant levels and goals in the greater context of detection capability and evaluate the extent to which different aspects of the regulatory apparatus (i.e., analytical capability, cost-benefit analysis, analysis of competing risks, and available toxicological data) influence the regulatory process. Our findings do not support the claim that decreases in detection limit lead to more stringent regulation in the context of drinking water regulation in the United States. Further, based on our analysis of the National Primary Drinking Water Regulation and existing United States Environmental Protection Agency approaches to establishing the practical quantifiable level, we conclude that in the absence of changes to the underlying toxicological model, regulatory revision is unlikely.

3.1 <u>Introduction</u>

Relevant toxicological models are generally understood to be the dominant influence on environmental exposure standards, including drinking water quality criteria. However, minimum detection capability serves as the lower limit below which regulation is not possible. In this way, ability to regulate is dependent on ability to detect, and advances in analytical chemistry allow for more stringent regulations.

We evaluate the vanishing zero claim with respect to drinking water regulation through the following steps:

1. Establishing a reregulation mechanism: We outline a regulatory mechanism based on the relationships between analytical capability, toxicological models, and cost benefit analysis and delineate the mechanism by which increased detection ability would bring about more stringent regulations;

2. Grouping currently regulated contaminants: We organize a subset of relevant contaminants, as outlined by the mechanism established in ref 1, according to relationships between final regulations, analytical capability, and toxicological

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[3] This chapter is the basis for a journal paper with the same title co-authored by Ketra Schmitt, published in *Environmental Science & Technology*, Vol. 44 (21), 2010.

goals to identify which regulated contaminants are theoretically most susceptible to reregulation on the basis of improved detection ability;

3. Assessment of the significance of detection capability in reregulation mechanism: We describe the ongoing regulation and reregulation assessment process at US EPA and evaluate the extent to which improved detection ability is influencing the regulatory apparatus, in terms of the delineated contaminant groups and mechanisms;

4. Review of real past regulatory changes: We document previous regulatory revisions against a timeline of analytical capability and establish the role analytical advances have historically played in triggering regulatory revisions; and

5. Significance of detection capability in emerging contaminants: We examine the process by which US EPA identifies new regulatory opportunities and assess the limitations imposed by, and overall significance of, detection ability.

To our knowledge, this study is the only policy analysis to date focusing on the mechanism by which changes are made in contaminant regulation, or the significance of analytical advances in triggering reregulation, targeted at a specific regulatory apparatus. This study considers drinking water regulation only in the United States, the only country whose regulatory apparatus our research has found to be the subject of documented claims of ‘the vanishing zero effect’ or ‘detection limit creep’. Indeed, a comparison of drinking water standards for 67 chemical contaminants promulgated variously by the US EPA, the World Health Organization, and the governments of Australia and Canada show that there is much room for drinking water standards in the U.S. to ‘creep’ downward: only 28 of the 67 contaminants targeted by this international comparison are currently regulated by the US EPA, and of these 28 contaminants, the standard enforced by the US EPA is greater than the smallest value in the comparison by at least 1 order of magnitude for 15 of them (Boyd 2006). Augmenting the findings of this study with the remaining contaminants whose regulations are currently limited in the U.S. by detection ability (a total of 24), we find that nine of them are greater by at least an order of magnitude than the guideline value of at least one other country in the comparison.

3.2 **Background**

The Safe Drinking Water Act (SDWA) empowers US EPA to regulate drinking water quality in the context of cost-benefit analysis and available treatment technology and toxicological data (AWWA 1999). It should be noted that the role of cost-benefit analysis in the many statutes administered by US EPA is not uniform: economic analysis weighing benefits against cost-related factors is specifically limited for a number of statutes under US EPA’s jurisdiction (Morgenstern 1994). In contrast, the standards US

EPA sets within the framework of the SDWA are the result of economic analysis that balances cost of compliance and expected benefits in the greater context of analytical and treatment feasibility (AWWA 1999).

The National Primary Drinking Water Regulations (NPDWR) are the end results of toxicological, technical, and economic assessment and constitute the list of legally enforceable standards promulgated under the SDWA. The NPDWR target a total of 84 water contaminants as well as three indicators. Since 1998, US EPA has published contaminant candidate lists (CCL) every five years to track potential targets for regulation under NPDWR (US EPA 2009b). From the CCLs, US EPA is required to make a regulatory determination on five contaminants every five years, on the basis of the danger posed by different chemical contaminants, the frequency and extent of their occurrence in public water systems, and the potential of a regulation to protect public health (NRC 1999).

US EPA relies on two parameters to communicate the recognized detection capabilities for each of the contaminants it regulates: the method detection limit (MDL) and the practical quantification limit (PQL). For a thorough discussion of the operational definitions of each of these parameters, the reader is referred to Gibbons's analysis (1994). However, we summarize MDL as the lowest concentration at which the presence of a given analyte can be confirmed (differentiated from a zero concentration) with a given analytical method to 99% certainty and the PQL as the lowest concentration of an analyte that can be reliably measured and differentiated from other, nonzero concentrations. The PQL can be determined in two ways. First, by real laboratory performance, where data are available, the PQL corresponds to the lowest concentration accurately measured by 75% of US EPA regional and state laboratories (US EPA 2006). Alternatively, the PQL can be calculated by multiplying the MDL by 5 or 10, according to the uncertainty or level of conservativeness required (US EPA 2006); whereas the MDL is determined experimentally and is specific to the operator and instrument at the time of analysis, the PQL is calculated by US EPA as a measure of typical expected lab performance (Eaton et al. 1998). US EPA therefore considers the PQL, rather than the MDL, the lowest level to which it is technically feasible to regulate. US EPA specifies the MDL required of tests performed on drinking water contaminants (CFR title 40) but uses the PQL as a measure of regulatory feasibility. Hamilton et al. (2003) compare drinking water standards to the MDL data, but not to PQL data, as a comment on the role of analytical capability in pesticide regulation.

The third quantity of interest for this analysis is the maximum contaminant level goals (MCLG). MCLGs are the concentrations of contaminants in drinking water that are not expected to have an adverse effect on human health (AWWA 1999). US EPA sets MCLG values based on the available evidence of carcinogenicity, according to weight of evidence categories (US EPA 2009c). For known or probable human carcinogens, the

MCLG is zero, because US EPA considers that no dose of carcinogenic contaminants will present zero risk, unless there is specific evidence to the contrary for a given contaminant (US EPA 1998). The fourth quantity of interest is the maximum contaminant level (MCL), which is the numerical value of the regulation US EPA sets for drinking water contaminants. The MCL must be set as close to the MCLG as feasible, taking into account cost and technical factors.

A 1996 amendment to the SDWA requires US EPA to review existing NPDWR and make revisions, where appropriate (US EPA 2003m). An integral part of this process is an analytical feasibility review to identify where US EPA is technically able to revise the PQL and, in some cases, the MCL. Reviews conducted in 2003 and 2009 assessed all but three chemical contaminants on the NPDWR in terms of the feasibility of lowering their PQL. The 2003 review did not target all the contaminants on the NPDWR but rather two subsets: 1) those whose MCL is limited by analytical feasibility (MCLG < PQL) MCL) and 2) those contaminants considered by US EPA as being likely or possible targets of toxicological review determining the appropriateness of lowering the MCLG (another component of the six-year review). The 2003 review targeted a total of 40 chemical contaminants, including 22 of the 24 contaminants where MCLG < PQL) MCL, and 18 contaminants regulated less stringently than technically possible but that were likely to undergo a toxicological review of the MCLG. The 2009 review re-examined all but one of these 40 contaminants along with 27 others, including the two remaining contaminants where MCLG < PQL) MCL. The 2003 Review targeted almost all of the contaminants of immediate relevance to our analysis and presented more detailed information on improvements in analytical sensitivity and so is more relevant to our study than the 2009 Review.

3.2 Findings

3.2.1 Establishing a mechanism

We inferred the limiting effect of different parts of the regulatory apparatus by organizing contaminants listed in the NPDWR according to the relationships between the public health goal (MCLG), regulation (MCL), and detection capability (PQL), as illustrated in Figure 3.1. Figure 3.1 shows the two main possible cases: either the regulation equals the public health goal (Relationship 1) or the regulation is less protective than the public health goal. A regulation can be less protective than the public health goal where it is limited by detection ability (Relationship 2) or where limited by cost-benefit analysis at some point above detection capability (Relationship 3).

If the regulation is fully protective of public health (Relationship 1), it is only likely to be revised if there is a change to the underlying toxicological model, not as a result of increased detection ability, because no public health motivation exists for a stricter regulation. If limited by cost-benefit analysis (Relationship 3), improved detection ability reflected by a lowered PQL would not be immediately relevant. US EPA would already have decided that it is not economically feasible to regulate to the level technically possible. If, however, the regulation is limited by analytical capability (Relationship 2), a change to the PQL could precipitate a change to the regulation promulgated by US EPA. Therefore, Relationship 2 is the most relevant to our analysis as it includes the contaminants susceptible to reregulation as a result of improved detection capability.

Ferguson et al. (2007) identified all regulated drinking water contaminants where MCLG < PQL, including arsenic, the subject of their study. They used the observation that MCLG < PQL to conclude that lowered detection ability may facilitate a lowered MCL. However, the authors did not comment on the relationship between the PQL and the MCL itself: the current PQL for arsenic is lower than the MCL, meaning that the regulation is limited not by technical feasibility but by cost-benefit analysis. While the PQL may stand to be lowered, we cannot conclude that it is likely to have an impact on the regulation set by US EPA (US EPA 2003m). Indeed, US EPA stated specifically that the arsenic rule protects public health to the extent that the benefits justify the costs (US EPA 2001a).

Recall that US EPA's study evaluated the technical feasibility of lowering regulations for 1) those contaminants currently regulated to the limit of detection (MCL) PQL, Relationship 2) and 2) where a possible revision to the toxicological model might change the MCLG, hence perhaps the MCL. While advances in analytical capability could facilitate a lower MCL (more stringent regulation) for either group, these advances can actively influence a change in MCL only where detection capabilities are the dominant factor (Relationship 2). We previously identified Relationship 2 contaminants to be the most sensitive to reregulation on the basis of improved detection capabilities. Assuming that such reregulation occurs, the resulting relationship between the new regulation (MCL'), the new detection limit (PQL'), and the public health goal (MCLG) would imply reclassification of the contaminant under Relationship 2, or 3, as illustrated in Figure 3.2. The new MCL (MCL') could be set equal to the new PQL (PQL') if it is economical to do so (Relationship 2 remains). If US EPA decides it is not economical to regulate to this level, whether PQL' is higher or lower than the public health goal (MCLG), MCL' would only be regulated to that cost-effective level (Relationship 3 takes over). If PQL' is lower than or equal to the public health goal and it is economical to regulate to that level, MCL' will only be set as low as the public health goal, as there would be no toxicological need to regulate to a lower level (Relationship 1 takes over).

We should note that many of the contaminants currently classified under Relationship 2 are carcinogenic compounds, with an MCLG ≠ 0. For these contaminants, it would not be possible for Relationship 1 to take over, since it is impossible to detect and therefore regulate to zero.

For all contaminants, detection capability (PQL), cost-benefit analysis, and toxicological knowledge (MCLG) all represent some value below which the MCL will not be set.

Figure 3.3 maps two conceptions of how drinking water regulations (MCL) change in response to improved detection capability (lower PQL) for contaminants whose regulatory levels are constrained by detection capability (Relationship 2). The blue line illustrates the mechanism we have outlined, whereby the MCL may be revised to follow the PQL until it reaches the strictest MCL justified by cost-benefit analysis, which we have represented in the figure as MCLCB (Relationship 3 takes over), or until it reaches the public health goal (MCLG) beyond which there would be no need to regulate (Relationship 1 takes over). This contrasts with the red line that represents unconstrained regulation along with the detection capabilities at levels closer and closer to zero as might be suggested by the 'vanishing zero' effect. Depending on the contaminant, MCLCB may be greater, less than, or equal to the MCLG (the variable relationship between MCLCB and MCLG is represented in Figure 3.3a,b by double-headed arrows). For most carcinogenic contaminants, the MCLG is equal to zero (in the absence of evidence that indicates the existence of a safe nonzero concentration). For contaminants that are the subject of competing risk analysis, such as between risk of illness from waterborne pathogens and cancer risk from disinfection by-products, the MCL is calculated in consideration of these competing risks.

Figure 3.3b reproduces the graph of the mechanism we outlined above, identifying where Relationships 1, 2, and 3 fit in. For a given contaminant, the MCLCB may be greater or lower than the MCLG. If MCLCB > MCLG, as depicted in Figure 3.3a,b, we would expect cost-benefit analysis to limit further regulation (Relationship 3 takes over) before the public health goal. If that relationship is reversed (MCLG > MCLCB), as indicated by the hashed arrows, then we would expect the public health goal to limit further regulation (Relationship1 takes over) before cost-benefit analysis.

3.2.2 Grouping currently regulated contaminants

In the 2003 and 2009 analytical feasibility reviews, US EPA identified a total of 24 contaminants where the MCL is currently tied to the PQL, representing a regulatory mechanism that directly connects detection capability with drinking water standard (11, 12). Generally speaking, these are the known or probable human carcinogens to which US EPA considers there to be no safe exposure. According to the regulatory mechanism outlined above, the MCLs of these 24 are not limited by cost-benefit analysis but by current detection abilities. This is also the case for two contaminants to which US EPA does consider there to be a safe nonzero exposure, that is, for which MCLG ≠ 0 (thallium and 1,1,2-trichloroethane). The remaining contaminants regulated to a value higher (less stringent) than the value possible with current analytical methods are regulated to the limit deemed necessary by available toxicological data. Thus, improvements in detection ability (decreases in PQL) would not necessarily lead to more stringent regulation. Rather, cost-benefit analysis could then prevent further strengthening of the regulation. A similar mechanism could facilitate the reregulation to a higher (less stringent) level of contaminants currently regulated to their MCLG if found not to be accurately represented by a linear dose-response profile (i.e., whose MCLG is re-evaluated above the PQL as the result of a toxicological or health review).

However, for contaminants where the MCL is currently tied to the PQL and for which improved detection capability facilitates a lower PQL, there is room for reregulation on the basis of improved analytical methods.

3.2.3 Assessment of the significance of detection capability in re-regulation mechanism

Since the 2003 analytical feasibility review aimed to determine whether data on laboratory performance can be used to reassess the PQL of different contaminants (hence creating the possibility of a reassessment of the MCL), we assessed correlation between an overall improvement of detection capabilities and a call to reassess the PQL for two subgroups of contaminants: MCLG < PQL ≠ MCL (Relationship 2) and PQL < MCL, where the MCLG might be subject to revision (a subset of Relationships 1 or 3) (US EPA 2003m). Therefore, US EPA assessed the practical feasibility of lowering the PQL of a large group of contaminants, some of which are classified under Relationship 2. Recall that this is the group for which the PQL is most relevant, and for which a revised PQL could logically lead to a revised MCL if cost-benefit analysis does not stop this from happening. The 2009 analytical feasibility review examined the same contaminants, plus two more whose regulation is contingent on detection ability (Relationship 2), and most of the remaining chemical contaminants not likely to undergo an MCLG revision and therefore not targeted by the 2003 review.

We organized the 67 chemical contaminants targeted by the 2003 and 2009 analytical feasibility reviews according to the relationship between overall improvement in the sensitivity of methods since promulgation and the conclusion that a PQL re-evaluation is in order (11, 12). This categorization was facilitated for contaminants targeted by the 2003 review by the detailed information available on the improvement of analytical methods since time of promulgation. For the 27 contaminants targeted by the 2009 review and not targeted by the 2003 review, only improvements in analytical sensitivity having occurred between 2000 and 2007 were considered in the supporting documentation. It was therefore not possible to perform a parallel classification for these contaminants. Readers interested in the classification of each contaminant are referred to Table 3.1. For the 40 contaminants targeted by the 2003 review, we compared the distribution of conclusions as to the appropriateness of a PQL readjustment between two main groups of contaminants: those whose regulations are currently limited by detection capability (Relationship 2) and those currently limited by toxicological model or cost-benefit analysis (Relationships 1 or 3). For each of these two groups, we consider the four permutations of two yes/no possibilities whether there has been an overall improvement in detection abilities since promulgation by US EPA, and whether US EPA concluded that a PQL reassessment may be appropriate. The distribution of contaminants within these four cases for the different Relationships will be discussed below. With these two groups and four cases, we can assess the correlation between advances in analytical capabilities and the PQL reassessment conclusion both for the group whose MCLs were identified as potentially susceptible to the influence of improved detection abilities (Relationship 2) and the group whose MCLs should be a function only of MCLG and cost benefit analysis (Relationship 1 or 3, respectively). Contaminants targeted by the 2003 analytical feasibility study are about evenly distributed between Relationship 2 (total of 22) and Relationships 1 or 3 (total of 18). Also, the distribution of contaminants among the four cases is roughly equal between Relationships 2 and Relationships 1 or 3. For both groups of contaminants, US EPA's conclusion that a PQL re-evaluation is likely necessary was more common for those contaminants where no real improvement in sensitivity had occurred since promulgation: for Relationship 2 (MCL controlled by PQL), 15 contaminants were identified as candidates for a PQL revision. Out of these, 12 had had no overall improvement in the sensitivity of methods, while only three had had such an improvement. Similarly, 10 contaminants in the Relationship 1 or 3 group were identified as PQL re-evaluation candidates, with 8 having seen no improvement in detection capabilities. Examining specifically those contaminants having seen an improvement in detection ability, we see that for Relationship 2, three were identified as PQL re-evaluation candidates, and two were not. Similarly, under Relationship 1 or 3, two contaminants having seen improvements in detection abilities were identified as PQL re-evaluation candidates and two were not. Improvement in detection capability is therefore not a useful indicator for predictions of PQL reassessment. An identical analysis based on the 2009 review was limited by the fact that only information on improvements in analytical sensitivity between 2000 and 2007 was available. For these contaminants, we also found that improvement in detection ability was not a useful indicator for predictions of PQL reassessment, but our sample of contaminants having undergone improvements in analytical ability was likely underrepresented. These comparisons are tabulated in Table 3.2.

US EPA identified a reason why it has so far been unsuccessful at establishing lower PQLs for contaminants targeted by the Six-Year Reviews (2003, 2009). US EPA determines the PQL by sending samples spiked with different concentrations of different contaminants to real laboratories and plotting the percentage of laboratories 'passing' vs. concentration tested. Laboratories that pass are those that accurately quantify the spike sample within specified acceptance limits. US EPA defines PQL as the concentration at which 75% of laboratories successfully detect the concentration in the spike sample. US

EPA refers to these as its ‘Water Studies’. Where laboratory performance testing yielded consistently high results, determining the concentration at which 75% of laboratories would pass was not possible, limiting US EPA’s ability to make an assessment of what might be a more accurate PQL. The analytical feasibility support document relied on real laboratory performance data to evaluate the possibility of revising the PQL for different contaminants. In several instances, the water studies tested the detection capabilities of laboratories at concentrations higher than the concentrations of interest and thus were of limited use. This could represent a significant practical barrier to the mechanism connecting advances in detection capabilities with PQL revisions. If the Water Study data US EPA used to reassess PQLs reflected performance at concentrations closer to the real detection limits, there may have been greater basis for PQL re-evaluation.

We note that US EPA has, for most contaminants, approved several analytical methods with detection limits that differ a great deal: up to several orders of magnitude in some cases. The use of analytical methods with higher detection limits would seem to short-circuit the process by which PQLs are re-evaluated: laboratory performance is not as good as is technically possible in accurately measuring low concentrations of different contaminants. However, the fact that Water Studies systematically test laboratory performance at concentrations higher than current PQL pre-empts the relevance of the problem of unnecessarily high detection limits. Until the Water Studies accumulate performance data at lower concentrations of drinking water contaminants, there is no basis to say that too-high detection limits in US EPA laboratories are impeding any necessary adjustments to contaminant PQLs.

3.2.4 Significance of detection ability to emerging contaminants

We compiled the previous regulatory revisions made to the NPDWR and assessed the role of detection capability in facilitating or precipitating the change for each regulatory revision. A total of 15 drinking water standards have been revised since they were first promulgated by US EPA or its predecessors (US EPA 2001b). Out of these, six have been lowered while seven have been raised. Additionally, the nickel regulation was remanded, and the lead regulation was changed from an MCL to a required treatment technique[4] (in addition to other requirements of the broader Lead and Copper Rule). Out of the 15 contaminants whose numerical standards were changed (including consideration of standards that predate the SDWA), seven have actually increased (become less stringent) since originally promulgated, while one other has been remanded. The arsenic standard was lowered, not to the PQL, but rather, to the MCLCB, as discussed earlier; this is also the case for cadmium, methoxychlor, and 2,4-D. The MCL for lead was

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[4] If the 90th percentile result of lead test results exceeds the action level of 0.015 mg/L, then additional monitoring, corrosion control treatment, source water monitoring/treatment, public education and/or lead service line replacement will be required based on system size and type.

replaced with a required treatment technique. Available data on historical detection capabilities for these 12 contaminants suggest that regulation to a lower value is possible but that incomplete toxicological data or economic realities have resulted in a higher value. For the remaining two (lindane and toxaphene), a thorough search through regulatory support documents identified no evidence explicitly tying the changes to advances in detection capabilities, although we notice that their MCLs are both equal to their PQLs. Table 3.3, in the Supporting Information, presents a summary of previous regulatory revisions with comments on historical detection capabilities.

3.2.5 Review of real past regulatory changes

Turning our attention from revisions to existing standards, we can examine the role detection ability plays in the implementation of new regulations.

US EPA uses frequency of detections and concentrations detected to make a judgment on whether regulation would protect public health for regulatory determinations (US EPA 2009d). We discussed the practice of regulating to the detection limit when this is the lowest level obtainable. However, if a candidate contaminant is only present in drinking water samples below the detection limit, a regulation would have limited potential to protect public health, likely precluding promulgation of an MCL. Advances in detection abilities could change this assessment. For potential targets of regulation, common occurrence of nondetects, or very low median concentrations of contaminants suggest detection ability might play a crucial role in the regulatory process. Contaminants not regulated today may be regulated when we become able to detect them.

We reviewed the screening support document used to filter potential contaminants on the third contaminant candidate list (CCL3) to assess the significance of limitations attributable to detection capabilities. To assemble a preliminary CCL, US EPA starts by assigning the contaminants to “toxicity categories” 1 (most toxic) to 5 (least toxic), on the basis of toxicological data and weight of evidence narratives (US EPA 2009d). It then considers the median concentrations found of the contaminants in representative finished drinking water samples. The more toxic a contaminant, the lower the median concentration detected needs to be before it is considered a potential regulatory target. Out of the 107 chemicals arrayed in terms of toxicity category and median concentration, only one was in the lowest concentration range (0 to <0.1 μg), although, this chemical is in toxicity category 1 (most toxic). The bulk of potentially regulated contaminants are present in concentrations substantially higher than the detection limit. Thus, US EPA is already able to promulgate MCLs that would have a measurable effect on the concentrations found in finished drinking water. We cannot therefore conclude that detection capabilities are preventing promulgation of new drinking water regulations.

Future improved detection capabilities are unlikely to lead to regulation of currently unregulated contaminants. Where detection ability prevents regulation of contaminants down to the level fully protective of public health or down to the lowest level that would be cost-effective, US EPA would presumably resort to its practice of setting the MCL to the PQL (Relationship 2). Candidate contaminants are as constrained by detection ability as the large number of currently regulated contaminants already classified under Relationship 2.

3.3 Discussion

We outlined a mechanism by which improved detection capabilities would lead to lower regulations but found little evidence showing that this mechanism has had or will have a significant effect on the overall trend of regulation revisions as compared to the other factors US EPA considers in its regulatory determinations.

Drinking water contaminants can be grouped according to the relationship between the associated public health goal (MCLG), recognized detection capability (PQL), and regulatory limit (MCL) in order to identify which are most susceptible to reregulation on the basis of improved detection, which are limited by cost-benefit analysis, and which are limited by the public health goal. Drinking water contaminants whose regulations are now equal to the recognized detection capability ($MCL \neq PQL$) are the most vulnerable to reregulation on the basis of the mechanism we identified, but there are several reasons this does not tend to occur.

US EPA is required to demonstrate that the costs of regulation will be justified by the benefits of new or strengthened regulation; this severely limits the likelihood of regulations departing from justifiable levels. Additionally, the mechanism we identified by which lower MDLs would lead to lower recognized PQLs and potentially stricter MCLs is short-circuited by the lack of widespread implementation of more sensitive methods at individual laboratories. Laboratory freedom to choose among approved methods and their predominant tendency not to use the most sensitive available seems to suggest that laboratory performance falls short of what is technically possible.

Both laboratory underperformance, in terms of adopting more sensitive methods, and laboratory overperformance, with respect to the Water Studies, limit US EPA's ability to take a position on the feasibility of lowering PQLs and hence possibly lowering MCLs. However, it is hard to predict how the conclusions regarding PQL adjustment feasibility would change as a function of improved assessment methods on the part of US EPA.

Indeed, the difficulties associated with reregulating as a function of improved detection capabilities and/or assessing the real improvement of detection capabilities over time is echoed by the lack of precedent for this kind of revision.

Within the data set we studied, we found no historical evidence for the claim that improvements in analytical capability alone lead to decreases in regulatory limits. Although we identify a mechanism making this possible, we note that US EPA's methods for enforcing laboratory standards and determining the PQL seriously limit the extent to which this would occur. As a consequence, there is good reason to believe that reductions

in regulatory levels of contaminants will continue to be driven by improvements in toxicological understanding of contaminant effect or by improvements in economic feasibility of contaminant removal.

We further conclude that the process by which new regulatory opportunities are explored is not limited by current detection capability, based on the high median concentrations of the bulk of the universe of drinking water contaminants relative to their limits of detection. US EPA is technically able to pass new regulations that are at least as protective as existing regulations with respect to the constraints imposed by limits of detection. Although we do not find the vanishing zero concept within the regulatory apparatus per se, we must note that increased detection capability undeniably facilitates regulatory debate of increasing numbers of contaminants. This distinction is well illustrated by perchlorate regulation efforts: an improved ability to detect contaminants triggered calls to regulate, but ongoing cost-benefit and other analysis has so far stalled those efforts. When regulation results from this circuitous but more plausible version of vanishing zero, this indicates that cost-benefit analysis has identified a meaningful opportunity for protection of public health. Although the concentrations of contaminants are venturing into the imperceptible, we can be confident the anticipated benefits would not be.

Finally, this study only evaluates the claim that improved detection limits in and of themselves can lead to more stringent regulation. In the context of environmentally regulated industry, the vanishing zero claim would be more directly linked to the regulatory apparatus underlying the Clean Water Act or the Clean Air Act, which impose more hotly contested emissions controls.

CHAPTER 4

Regulation of chemical drinking water contaminants: risks, benefits and the hunt for equality in the Canadian context

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Drinking water quality management in Canada is based primarily on the intervention of individual provinces and territories. This contrasts with the American and European approach of uniform, legally enforced regulation at the federal or super-federal level. The Canadian model has been widely criticized for the unequal level of regulation between provinces and territories and the passive role taken by the federal government. Notably, criticism from environmental advocacy groups has focused to a large extent on the management chemical contaminants, whereas research on water governance has focused largely on the potential for pathogenic contamination. This chapter bridges the gap between a body of literature on water governance and mainstream discourse on jurisdictional issues in drinking water management. We investigate the principal arguments for giving legal effect to federal drinking water guidelines for chemical contaminants, compare popular claims about water quality with available data and add context with perspectives on risk perception and general principles in drinking water management. We compare current and proposed regulatory frameworks in terms of their implications for protectiveness of public health, responsiveness to emerging contaminants and equity among populations. We explain why claims of regional inequalities in drinking water quality are so persistent in the Canadian context but show that increasing the responsibility of the federal government may not lead to improved overall public health outcomes. We outline strategies for federal, provincial and territorial actors to improve the current decentralized framework and minimize the practical difficulties inherent in decentralized regulation. We conclude by explaining how a decentralized approach to regulation, based on risk management and economics, is most likely to be successful in the Canadian context when provinces and territories are actively engaged in the exercise of allocating money to public health initiatives in proportion with local risk profiles. Throughout, we identify opportunities for data collection that would allow for more confident, definitive pronouncements on risks and benefits of different regulatory structures and strategies.

4.1 **Introduction**

A series of outbreaks in Canadian public water systems in the early 2000s has made drinking water quality an important topic in public discourse. Although unsafe drinking water has usually been a topic connected to the developing world, the widely publicized Walkterton, North Battleford and Kashechewan outbreaks alerted the public that even in Canada, clean water cannot be taken for granted. Several prominent environmental groups have been critical of drinking water quality management in Canada (Sierra Legal

Defence Fund 2001 and 2006; Boyd 2006; Ecojustice et al. 2006: hereafter collectively referred to as 'the drinking water reports'). These leading environmental groups focus on the chronically poor quality of First Nations water supplies, the inequality of drinking water standards among provinces and territories and, notably, the level of regulation of chemical contaminants in Canada versus other countries. They conclude variously that the federal government is not sufficiently involved in drinking water management, that Canada's decentralized regulatory model is inequitable and jeopardizes the health of its citizens and that the level of regulation in Canada is below that of other developed countries.

Substantial research, both academic and government, has been directed at questions of water governance in Canada over the past ten years. This research tends to focus on water treatment system management and source water protection while taking current jurisdictional division of responsibilities as a given. On the other hand, researchers in environmental geography have thoroughly explored jurisdictional issues of water management, but mainly with respect to source water protection and allocation. Academic literature on jurisdictional issues in drinking water legislation in Canada is rare, and the water quality issues discussed focus on pathogenic contamination. Environmental groups however raise legitimate questions about the benefits of centralization versus decentralization of drinking water regulation, analysis and decision-making notably with respect to chemical contaminants, not often discussed in literature on water governance.

This paper is the first to examine the division of responsibility for drinking water quality in Canada in the context of total risk abatement including the risk represented by chemical contaminants. It is also the first academic paper to address the specific claims of environmental groups with respect to greater centralization. It brings to light compelling economic arguments for improving the current decentralized model as opposed to instituting greater centralization and explains the persistence and resonance of calls for centralization in the Canadian context.

In this paper, we:

1. put the claims made by Canada's leading environmental groups in the context of Canada's unique social and political climate and thereby explain their resonance with the public;
2. evaluate centralized and decentralized frameworks for drinking water regulation in the context of risk management theory and practical challenges in the Canadian context; and
3. explore the basis for inferring high risks from seemingly permissive regulation.

The paper closes by proposing areas of inquiry and data collection that would allow for more definite statements on the best way to rework Canadian drinking water policy in order to ensure long-term protection of public health. Particularly, it finds that the economic arguments in support of maintaining and improving decentralization are dependent on a demonstration that provinces and territories are actively engaged in risk abatement in other spheres of public policy.

4.2 Background

Water quality management became a topic of concern to many Canadians alarmed by widely publicized accounts of drinking water outbreaks in the early 2000s. Increased public awareness of the hazards of drinking water is reflected in the series of drinking water reports cited above. All these reports describe the current state of drinking water in Canada as unacceptable and criticise the nature and extent of the involvement of the federal government. As is generally the case (Slovic and Weber 2002), it has been extreme, isolated events like the outbreak in Walkerton, Ontario that have sparked the public's interest in safe drinking water. The outbreak in Walkerton was responsible for over 2,000 illnesses and seven deaths in a small community (O'Connor 2002). However, this accounted for less than 3% of the total illnesses and less than 8% of the total deaths thought to be attributable to contaminated drinking water in Canada every year (Environment Canada 2008). This demonstrates that large populations stand to benefit from a dialogue on the improved management of drinking water, even though this dialogue is moved along by low-probability, high-impact events like the outbreak in Walkerton.

The official policy of the federal government with respect to drinking water is known as the 'multi-barrier approach' (e.g. CCME 2004; Health Canada 2010) and aims to ensure access to safe drinking water through a system of regulation, monitoring, treatment and management. The Federal-Provincial-Territorial Committee on Drinking Water operates under Health Canada and publishes nonbinding guidelines for drinking water quality as well as related technical and toxicological support documents, guidelines for good practice, research reports, and other general information. Also, the federal government is responsible for drinking water quality on federal lands, including Indian Reserves. Otherwise, the ultimate responsibility under the law for water quality standards and other management measures falls to the provinces and territories.

In that way, Canada stands out from other western, industrialized countries for the relatively passive role taken by the federal government in matters of drinking water regulation. Canadian provinces and territories have a high level of autonomy in environmental regulation, an arrangement that has come about more by tradition than by legislation. Matters of the environment are not explicitly assigned to federal or provincial/territorial jurisdiction in the Constitution, but extrapolation from the treatment of natural resources has meant that it has been primarily the provinces and territories that are responsible for the regulation of the environment and settlement of conflict (Health Canada 2006). This arrangement has produced drinking water standards and management practices that vary a great deal between one province or territory and the next (Hill et al. 2008). Some critics have referred to this as a 'patchwork' of inequitable rules for drinking water management (Sierra Legal Defence Fund 2001 and 2006, Ecojustice et al. 2010) whereby some Canadians are protected by more stringent rules than others. The US and the European Union, on the other hand, have adopted federally (or, in the case of the EU, super-federally) mandated drinking water quality guidelines that ensure consistent standards across their respective territories.

Claims of inadequate federal intervention have also been based on the numerical value of maximum allowable concentrations (MAC) in Canada versus those of other countries; one study points out that Canada's guidelines are very often less stringent than those of the US, Australia or even the World Health Organization (WHO) (Boyd 2006). The investigation into the Walkerton tragedy suggests that the relative permissiveness of Canadian regulations might be attributable to the federal-provincial-territorial subcommittee's orientation towards consensus, which often leads to agreement on the 'lowest common denominator' of public protectiveness (O'Connor 2002). Although, off federal lands, the Guidelines are not legally binding, they serve as a guide to decision-making and undeniably have bearing on the standards adopted by provinces and territories. For example, Alberta has adopted the bulk of the federal Guidelines as its own legally binding standards.

Drinking water quality management can be divided into two branches: 1) strategies to protect against short-term outbreaks, primarily attributable to spikes in microbiological activity in water supplies, and 2) management of long-term risks, usually associated with chronic trace chemical contamination. The principles governing the management of short-term risks are relatively few and well understood: monitor microbiological activity and treat contamination with standard treatment methods such as chlorination and filtration. The regulatory framework typically focuses on mandatory reporting of analytical results for microbiological parameters or chlorine residual and operator training requirements. The nominal guideline values for microbiological contaminants are often zero, or a prescribed treatment efficacy, and water quality goals vary little among industrialized countries. In contrast, long-term risks are managed by setting maximum allowable levels of contamination by various chemicals. Regulatory agencies determine the level of exposure that is safe by consulting toxicological data and balance these considerations with the cost and potential benefit of regulation. Extensive toxicological research, exposure characterizations and cost-benefit analysis govern the final numerical value that is assigned to these contaminants. As we will see, the higher the stakes and implications of the regulation, the more extensive and protracted the analysis. Competing opportunities for public investment, frequency of contaminant occurrence and other local factors play important roles in regulatory decision-making that create potentially large differences in regulatory limits.

Almost all known cases of illness and death due to drinking water contamination are attributable to breakdowns in these processes. Conversely, illness and death due to chemical contamination are extremely hard to trace back to drinking water on an individual level. This is due to the very small numbers of people that typically get sick, the long exposure periods that are typically necessary to cause a health response in exposed populations and the fact that the resulting diseases (notably cancer) cannot be categorized on an individual level according to cause. Correspondingly, the bulk of academic literature and government analysis on drinking water in Canada focuses on ensuring access to water free of the pathogens associated with immediately identifiable short-term illness and death. Notwithstanding this focus on the part of drinking water researchers and managers, environmental advocacy groups have devoted a high proportion of their attention to the management of chemical contaminants. This raises three interesting questions, in line with the three objectives of this paper: 1) why does the Canadian regulatory model for chemical drinking water contaminants represent such a liability in the eyes of environmental advocates? 2) Would a centralized approach lead to better water quality outcomes, and would it be more equitable? and 3) Are environmental advocates justified in inferring high risk from low regulation of chemical contaminants?

4.3 Perceptions of risk and inequality as a driver for regulatory reform

The outbreaks that capture public attention are not primarily related to the system of drinking water guidelines that establish maximum allowable exposures for a variety of contaminants as a function of cost benefit analysis, but rather, to breakdowns in mechanisms that are supposed to prevent exposure to a select group of microbiological parameters. In other words, short-term risks, rather than long-term, were behind Canada's drinking water tragedies. However, public discussion of water quality management conflates these two types of risk. The drinking water reports bring public attention to the broader range of federal guidelines which they deem not adequately protective of public health for two reasons: they are not legally binding and they are in many cases less stringent than those of at least one other country. The extent to which claims of unsafe drinking water resonate with the public are highly consistent with research in risk perception. Pilisuk et al. (1987) found that drinking water tops the list of public concern for technological hazards with 79.1 % of respondents identifying as 'very concerned' and a further 15.1 'somewhat concerned'. For comparison, 54.9% of respondents in the Pilisuk study described themselves as 'very concerned' about nuclear war. Concerns over abstract risks in drinking water are at the surface of the popular consciousness and may be easily triggered by claims of lax protection of water resources supported by easily relatable examples of regulatory failures.

Regional egalitarianism is a defining characteristic of the Canadian social context. Public polls indicate that Canadians are highly supportive of egalitarianism, that is, that income and opportunity should be independent of ability (Aalberg 2003). Canada's system of equalization payments is a widely recognized symbol of inter-regional egalitarianism, in place since the 1950s and entrenched in the 1982 constitution, aiming to harmonize the quality of social services between richer and poorer provinces (Kellock and LeRoy 2007; Kasoff and Drennen 2008). These expectations of an egalitarianism that transcends economic realities are almost universal and, particularly in discussions of public health, unquestioned, with one study describing it as the 'mantra' of the Canadian context (Giacomini et al. 2004). Indeed, regionalism, as important as national unity and social welfare, has dominated political discourse and conceptions of representativeness and equality since Confederation. This contrasts starkly with, for example, the United States, where sex, race and ideology are the prevailing politically relevant categories (Malcolmson and Myers 2009). In the next sections, we examine motivations for addressing water quality management on a provincial level. Environmental advocacy groups argue that regional regulatory disparities imply regional injustice; a claim that resonates strongly with Canadians.

The drinking water reports have been widely covered in the media and rank highly among internet search results for drinking water quality in Canada. Most other sources of Canadian water quality information are provided by the federal or provincial governments. These sources often fail to provide real data that laypeople could use to create informed opinions on water quality risk. Explicit references to events in the public consciousness (e.g. the Walkerton outbreak) are uncommon in the government literature reviewed. Other literature on the realities of drinking water in Canada is rare. Indeed, the drinking water reports provide a detailed and accessible overview of drinking water policies and reference current events of which an informed public would be aware. Although the focus of the reports varies, they establish a body of references that conclude Canada's drinking water regulatory apparatus is flawed and that the risks are great. Because the drinking water reports represent many trusted names in environmental stewardship, greatly surpass other sources of information in terms of accessibility of information and agree with the risk implied by well-known and widely reported outbreaks, they likely play a large role in shaping the policy opinions of their readership.

4.4 Regulation and water quality: limited basis for inference

Popular discussion of drinking water quality in Canada, as exemplified by the drinking water reports, very often uses regulatory stringency as an analogue for water quality: high risk is inferred from lax legislation. This central claim is supported by demonstrating variously that the Canada-wide standards for chemical contaminants are weak compared to those of other countries, that they do not have the force of law, and that many provinces have not adopted strong legislation to protect drinking water from chemical contamination. The argument that high-risk drinking water can be inferred from lax regulation is problematic in three ways: 1) regulatory stringency (or laxness) is a concept that has not yet been quantified and therefore cannot be used to rank jurisdictions in order of their protectiveness of public health; 2) there is no established multi-attribute framework for the risk of drinking water that would allow it to be similarly ranked in order of risk posed; and 3) beyond these barriers of analysis, there is no logical basis to infer high risk from low regulation or vice-versa.

The difficulty of classifying drinking water according to total risk is attributable to the lack of framework for multi-attribute water quality (Schmitt and Calder 2011). Chemical drinking water contaminants carry diverse and disparate risks: long-term cancer, short-term disability, reproductive difficulties and more. The difficulty of comparing the magnitude of these disparate risks is compounded by the diverse susceptibilities of individuals constituting a population served by a common drinking water source. The lack of methodology for establishing a "risk score" that prevents us from ranking drinking water in terms of absolute quality is the fundamental barrier to making a similar ranking of regulatory stringency. However, regulatory stringency is not only defined by the total risk abated, but by other measures, such as operator qualification and supervision, water quality monitoring and source-water protection. The relationship between source water and drinking water quality is further complicated by the effectiveness of the 'barriers' between the two, notably, treatment, testing and operator qualification. For example, the failure of these barriers was responsible for the Walkerton outbreak (O'Connor 2002).

Figure 1 organizes source water and drinking water by risk, where 'high' indicates that the source waters are high-risk due to manmade or natural contamination, and 'low' indicates that the source waters are low-risk due to a low level of contamination. Figure 1 shows that high-risk source water results in high-risk drinking water when regulation is high but ineffective or when regulation is inadequate. When underlying risk is low, however, the underlying regulatory framework is less important; neither high- nor low-stringency nor effective nor ineffective regulation changes the risk of finished drinking water. A comparatively low level of regulation therefore does not imply high risk. Reasoning to the contrary ignores the probability that a given source water is low-risk to start with. Similarly, stringent regulation on paper might be improperly implemented, meaning that high regulation will not necessarily result in lower risk.

To fully understand the difficulty of inferring drinking water risk from the level of regulatory intervention, we refer to Figure 2, which presents all permutations of source water quality, regulatory stringency and regulatory effectiveness. Despite the important barriers to a more quantified analysis, we generally understand the current drinking water regulatory paradigm to be reactive rather than anticipatory, with drinking water regulations cropping up in a piecemeal way in response to emergent risks (Schmitt and

Calder 2011). Therefore, it seems more likely that a given dense regulatory landscape would imply an attempt at reducing a high source-water risk rather than a pre-emptive regulation of already-safe water. Therefore, it is no more logical to infer high risks as opposed to low risks from permissive regulations: they may be lax because of low risk, or there may be high risk on account of lax regulations. The level of protection mandated by policy against a *hazard* says nothing about the magnitude of the associated underlying *risk* the policy is addressing. It is tempting to compare regulations (or any other measure to protect public health or wellbeing for drinking water or otherwise) between different jurisdictions and make inferences about the risks faced by those populations. However, given the limited data and tools for this type of analysis, the abundance of specific claims of high risk are likely to be premature.

This structured conception of relative risk and stringency and the difficulties of rational comparison deserve individual attention in a separate paper. However, the difficulties inherent in doing this are well illustrated by Boyd's *The Water We Drink*, published by the David Suzuki Foundation. In this report, we find a partial list of drinking water contaminants and the guideline (or regulated) values published by the governments of Australia, Canada and the United States, and by the World Health Organization (WHO). Boyd explains that 'there are 55 contaminants for which Canada has weaker guidelines for the protection of drinking water quality than at least one other jurisdiction or the World Health Organization recommendation.' He goes on to cite rankings of Canada and other governments and the WHO in terms of level of regulation for different contaminants. He paints a grim picture of the level of federal engagement in drinking water protection based on the observation that for the subset of contaminants examined, the Canadian guidelines are often not the most stringent value. This reasoning does not consider the magnitude of different provincial regulations that have the force of law in any case. In addition, we note that although Canada does not compare favourably in terms of level of numerical regulation for most contaminants, Canada has the highest number of regulated contaminants on that list. What are the implications for overall risk? Do the additional contaminants regulated compensate for the relatively low level of regulation of the entire list? What are the relative risk savings of other non-numerical water management practices? It is not possible to answer these questions without comprehensive exposure and toxicological data. We see that these simple comparisons quickly start asking more questions than they answer and are therefore perhaps better used to frame further enquiry into comparative risk analysis.

The analysis developed in this section is most robust in the context of chemical contaminants from established sources producing water of a predictable quality. In the real world, accidental and exceptional microbiological contamination of water supplies is a regular occurrence. The numerical values of drinking water standards are much less important in preventing and responding to these events than the diverse operational protocols and programs established to minimize short-term risk. Therefore, the argument that it is premature to infer risk from regulatory stringency and that regulatory stringency is too elusive a concept on which to judge jurisdictions applies only to unexceptional chemical contamination. This argument does not take away from demands that all drinking water systems be adequately prepared to handle unexpected pathogenic contamination that can befall any municipality, regardless of the normal water quality.

In probing the commentary on regulatory stringency and drinking water risk, we have uncovered two areas where further research would significantly improve the formulation and allocation of resources to public health policies: 1) the development of a multi-attribute measure of water quality and risk; and 2) the development of a measure of regulatory stringency. Likewise, the study of source water quality, regulatory response and resulting drinking water quality would allow us to identify 1) which management policies are effective in achieving public health goals; and 2) the likely future demands of the regulatory apparatus in an age of falling detection limits and densification of urban activity around receiving waters. Data on contaminant occurrence in source water and human exposure via drinking water are disappointingly rare and disorganized in Canada. Individual municipalities may publish results from water quality monitoring undertaken in compliance with relevant regulations, but data availability and management issues make it very difficult to compile the information and draw conclusions about drinking water quality characteristics on a national or even provincial basis. At the time of publication, no comprehensive drinking water contaminant exposure data has been published. Until the data necessary to make meaningful risk assessments on a large scale is available, our ability to undertake this type of research, to formulate a logical policy response on a national level or evaluate the effectiveness of current initiatives is severely limited.

4.5 Legal status of drinking water guidelines and precautionary regulation

The precautionary principle is the guiding regulatory principle in Europe and states that potential contaminants should be demonstrated to be safe before being widely used. The US and Canada use cost-effectiveness analysis with legally binding regulation; however, this does not apply to Canada's non-binding drinking water guidelines. As a consequence, the federal government is currently able to post drinking water guideline quality values and standards for good practice at its own discretion, with little justification of the associated costs. It is up to the provinces to adopt (or not) the federal guidelines if it determines the benefits justify the costs. As we have seen, criticisms of the federal government's role in drinking water quality cite its failure to be sufficiently engaged in protecting public health. However, if federal guidelines gained legal effect, we would anticipate a surge of criticism from the regulated industry and some provincial and territorial governments with demands for proof that expected benefits absolutely justify costs associated with meeting the guideline values. While Canadians would gain a common baseline water quality, we anticipate that this baseline would be designed to withstand cost-benefit analysis; a framework in many ways incompatible with precautionary regulation. It is instructive to turn to the example of the US, whose centralized drinking water regulatory system resembles the desires of the critics of the Canadian federal government's regulatory approach to drinking water.

The US EPA regulates drinking water quality at the federal level on the basis of the Safe Drinking Water Act (SDWA), its amendments and other guidelines, such as executive orders treating the role of cost-benefit analysis within regulatory agencies. Since the regulations passed by EPA require mandatory compliance of all drinking water systems in the country, prior to regulation, EPA is required to demonstrate that a drinking water contaminant poses a threat to human health, that it occurs with frequency and at levels of concern to public health, and that regulation represents a meaningful opportunity for health risk reductions. Since 1998, EPA has made several regulatory determinations on compounds it has selected for evaluation from its Contaminant Candidate Lists (CCL), the system by which it identifies and screens possible new targets for regulation. In total, 17 chemical contaminants have undergone review in an effort to assess whether health benefits would justify regulatory costs. Of these 17, six are thought to be present in the drinking water of at least one million Americans in levels that exceed EPA's health reference level, while another four are thought to occur above the health reference level in the drinking water of between 100,000 and one million Americans. Still, EPA has decided against regulation for all 17 contaminants it has evaluated (Federal Register 2003 and 2008). While it has taken the EPA ten years to decide against regulation of 17 emerging drinking water contaminants, some of which affect large populations across the US, Health Canada has adopted guideline values for four of these, along with another six identified as candidates for regulation by EPA, but still pending evaluation. As always, it is up to provinces and territories to opt in to these standards. The slow pace at which drinking water regulation is decided upon or against in the US speaks mainly to the retarding effect of cost-benefit analysis, where reaction to a risk is delayed until EPA can justify why or why not it is appropriate to enforce new standards of water quality on publicly funded water systems across the country. The difficulty of this task is compounded by the nationwide variation in the relationship between costs and benefit of a hazard and a potential regulation, and the inherently subjective exercise of deciding whether it is more worthwhile to regulate all or none, and to what extent.

We can only speculate on how Canadian federal agencies would have reacted if the stakes were as they are in the US, that is, with these regulations having the force of law. Still, it is worth noting some similarities that may justify extrapolation to the Canadian context. Like the US, Canada has a well-developed, mandatory regulatory impact assessment (RIA) process that is entrenched in the regulatory culture (Volkery 2004). The basic requirements of new regulations were reaffirmed most recently in 1999 and invoke a separate Environmental Assessment of Policy (EAP) and, burdensomely, a demonstration that the benefits *outweigh* the costs (Government of Canada 2000). This language is more severe than the analogous US directives for cost-benefit analysis; since 1993 only a demonstration that benefits of proposed regulations *justify* costs is required (Morgenstern 1999). As we have seen, data on contaminant occurrence on a national level are lacking in Canada, but we would not expect to see more homogeneous quality than in the US; the primary causes of water quality differences among American states are climatic, hydrologic and demographic variations as well as regionalization of manufacturing and agriculture (USEPA 1999) phenomena that define the Canadian context as well.

Proponents of federal regulation tend to encourage allowing provinces and territories to enact stricter rules as they deem appropriate, so that the federal standards serve only as a baseline. This is technically the case in the US, where individual states are free to regulate any contaminants as long as the regulations are not lower than those promulgated by EPA. However, we note that only a minority of states (California, New Jersey and New York – representing about 20% of the US population) have numerical drinking water standards more stringent than those of EPA. The additional coverage provided by these states still leaves the majority of highly exposed populations without legal protection at the state level. Thus, while the goal of federal and state protection is to provide public safety, they are overlooked by both levels instead of being protected by both levels. To cite but one example, an estimated 10,000 to 781,000 Americans, almost entirely in states lacking sub-national drinking water regulation, are exposed to hexachlorobutadiene in their drinking water at levels above the health reference level (USEPA 2003). USEPA opted not to regulate hexachlorobutadiene because exposed populations were still too rare to justify a national rule.

It is hard to extrapolate to the Canadian context. While a strengthened federal role might justify disengagement on the part of provinces and territories from this expensive regulatory arena, the self-propelling nature of bureaucracies might promote a dual system wherever provincial and territorial authorities have established a drinking water regulatory system. Indeed, we might expect a further fragmentation of the regulatory landscape as some provinces and territories disengage entirely and others stay their course. While we have shown the requirement to conduct cost-benefit analysis slows responsiveness to emerging contaminants, we are not arguing for less cost-benefit analysis or a more precautionary regulatory culture. Indeed, increased precaution in drinking water regulation would mean some other benefit foregone. As we will see, there are competing opportunities for risk reduction and, without cost-benefit analysis, we would have no way of ensuring we are allocating money rationally. The facilitation of individual decision-making should be a major accomplishment of the Canadian system of publishing extensive nonbinding guidelines, notwithstanding the lack of proof that the guidelines would be cost-effective or rational instruments of policy. Indeed, individuals should be as informed as possible about relative risks and free to spend more to avoid risk than what would be economically tenable regulation. However, this accomplishment is undermined by the consensus-oriented approach that dilutes the protectiveness of guideline values to appeal to provinces and territories that do not want to appear out of step with federal norms.

4.6 Centralization, equity and water quality as a local issue

Government-sponsored reviews of regulatory practices since the outbreaks in the early 2000s have found several opportunities for improvement, but they have tended to criticize the provincial government procedures for monitoring and reporting, rather than inadequate regulation. The Walkerton report observes that the federal-provincial-territorial subcommittee on water quality, the working group responsible for the nonbinding guidelines set at the federal level, has the tendency to settle on the lowest common denominator (i.e. less protective regulations) owing to its consensus-oriented structure. A stakeholder may agree to a regulation only as protective of public health as provided by the amount of money that stakeholder is willing to invest. For example, the high guideline value for turbidity is reportedly attributable to the Yukon being unwilling to invest in lowering turbidity and simultaneously not willing to be in violation of the Canadian guideline values; the result is that the standard at the federal level is higher than it would have been (O'Connor 2002).

As this example indicates, the Canadian experience with national guidelines has already been one *where cost considerations of the few may prevail over water quality considerations of the many*; this reality is attenuated by the fact that the federal guidelines have no legal effect anyway. This contrasts with the American experience where regulation of emerging contaminants at the national level has been hampered by *cost considerations of the many prevailing over health considerations of the few* as well as health considerations based on emerging science. The burden of justifying a regulation everywhere it is applied is a major roadblock to national regulation of a primarily local issue, especially when legally binding. Real or emulated, centralized regulatory frameworks are subject to pressures that may lead to missed opportunities to protect public health. This can occur as in the Yukon by accommodating the objections of lagging sub-national jurisdictions, or as in the US, out of an inability to justify mandatory intervention across the country when only small populations would benefit, or when the benefits are uncertain. Although drinking water regulations would become more equitable if centralized, it might in fact be at the expense of public health *in terms of drinking water risk*. Conversely, removing the ability of provinces and territories to decide what level of drinking water intervention it can afford in consideration of other opportunities for risk reduction might in fact increase *overall risk* if the opportunity cost for a province or territory to meet federally mandated drinking water standards is high. Canada would make better use of the existing decentralized nature of its regulatory framework by basing health guidelines on the best available science, rather than weakening them to satisfy the interests of the provinces or territories.

In spite of the pressures facing EPA, we see that it ensures a baseline water quality to all Americans. Indeed, greater centralization in line with the US model would seem to represent progress towards equity among Canadians. Advocates of increased responsibility for drinking water regulation on the part of the federal government and a generally more centralized regulatory system cite equity among Canadians and surer protection of public health as the overall goals. Although centralizing rules and guidelines for drinking water quality management would help harmonize the risk attributed to drinking water contaminants, it might in some instances be at the expense of overall public health or wellbeing. Widely publicized debates over the regulation of arsenic and perchlorate in the US exemplify the great difficulty of calculating tradeoffs between cost factors and protectiveness of public health on a national level. This is particularly true when detection ability allows for quantification of compounds at level lower than the dose at which a health response has been measured or at levels unachievable by conventional treatment technologies (Calder and Schmitt 2010).

Risk from drinking water contaminants, whether of short-term outbreaks or long-term cancer incidence, is but one part of a greater risk profile that influences public policy decisions. Standardization of water quality standards across Canada is in effect a move towards risk equity, but *in terms of drinking water risk alone*. The greater picture of risk is very regional, varying even from one Canadian province or territory to another. For example, the car accident death rate per 100,000 people between 2000 and 2004 was 7.0 in Ontario, but 16.4 in the Yukon Territory (Ramage-Morin 2008). Obesity rates in 2004 varied between 18% for men and 20% for women in British Columbia and 33% for men and 35% for women in Newfoundland and Labrador (Tjepkema 2008). Heavy drinking is more prevalent in Atlantic Provinces and in the Territories than elsewhere in Canada (PHAC 2010). Probably, risks attributable to drinking water quality vary nationally as well, though there has been no comprehensive study on the subject, as we have discussed. Regardless, it is certain that drinking water represents relatively greater risk in some jurisdictions than in others. The variability of these other risks means that drinking water risks do not rank consistently in regulatory priorities from one jurisdiction to the next. These other risks may also warrant public investment. Therefore, the Canadian model of decentralized regulation allows provinces and territories to decide on risk reduction priorities. It provides the freedom to, for example, allow some drinking water risk in exchange for reducing car accident or obesity risk, which may represent a more cost-efficient use of public money. Since risks to the public are a local phenomenon, coordinating risk reducing investments at a national or super-national level is likely to be ineffective. Spending to diffuse drinking water risk, while perhaps a good decision for one area, might be a bad decision for another, if the opportunity cost is high. This is the flipside of the issue we explored earlier: that centralized regulation may tend to default against regulation where it is not justifiable everywhere.

The water quality guidelines decided upon by the federal government are in effect the result of some cost-benefit analysis. It is a value judgement produced on the basis of national data that is not likely to be the same for every province, territory or even city. In view of the problems liable to be caused by giving them legal effect across Canada, the federal government might instead aim to provide the (more) neutral cost-benefit information to enable local regulators to decide how much intervention is appropriate in consideration of their other problems and opportunities for public investment. Throughout our discussion of drinking water realities in Canada, it is clear that we could make better judgements about water quality if we had access to better data. To some extent, this is due to a shortage of original research or efforts to knit existing information together. Some of this stems from a lack of data availability and general government transparency. Although nothing on the federal level is legally binding, Health Canada and associated working groups have published considerable amounts of advisory information (e.g. nonbinding standards) that is obviously tailored to the Canadian context. However, researchers and the public have access to very little information on the decision-making process used to inform final federal government publications. This contrasts with EPA that publishes all supporting documentation for its regulatory determinations and allows for reinterpretation by researchers and other stakeholders.

4.7 Practical considerations

So far, we have outlined compelling arguments suggesting that decentralization of responsibility for drinking water quality is the best way to ensure protection of public health: decentralization allows local populations to allocate resources rationally and proportionally in the context of competing opportunities for investment. However, the decision-making required within fully decentralized regulation would carry substantial ‘transaction costs’. Demanding that small agencies conduct cost-benefit analysis on the scale of what would be required to adopt a fully rational approach to investment in big-picture risk reduction is clearly not feasible. Ranking risks of death and of different short-term and long-term illness in terms of funding priorities and reconciling these with other public interests is too complex a task to demand from local governments. Analysis of tradeoffs and comparison of incommensurate risks rapidly becomes a subjective task, and it is not practical for this exercise to be repeated from start to finish in every province and territory.

The federal government therefore might make a more meaningful contribution to the protection of public health if it aimed more explicitly to facilitate decision-making on the part of provinces and territories. Information on risks, costs and benefits could be used by provincial, territorial, municipal or other regulators to determine the appropriate level of intervention in consideration of other opportunities for risk reduction. This would serve the dual purpose of promoting the protection of localized highly exposed populations where a centralized regulation might be unwarranted (as in the case of emerging contaminants not regulated by EPA) and accounting for the variable opportunity cost of drinking water risk reduction. Under this structure, for example, Yukon and Ontario could use the same data to justify respectively permissive and stringent standards for turbidity, a drinking water characteristic that is apparently a smaller spending priority in Yukon. The cost-benefit analysis would be done once, at the federal level, and the results would serve as the basis for provinces and territories to make rational decisions. Ontario already has binding regulations for a number of contaminants not regulated, for example, in the US, but it is not clear that the information on costs and benefits in the Ontarian context is available for other provinces and territories that might benefit from it.

Arguments for decentralization of drinking water regulation in Canada depend on the premise that provinces and territories would invest rationally in public health initiatives if the federal government did not intervene. That is, the arguments outlined in this paper are persuasive if centralization of drinking water regulation would deprive Canadian provinces and territories of opportunities to address bigger public health hazards with the funds they are instead made to allocate to drinking water. This is principally because of the entrenched role of government as a re-allocator of resources and decision-maker in the interest of equalizing public health outcomes. However, in countries where the role of the government in public health intervention is less certain, such as in the United States, the arguments are not necessarily so limited.

4.8 Native communities

Drinking water management in native communities poses many unique challenges, as much for the Federal government as for native leaders. The most obvious challenge native communities pose to drinking water decision makers is their status outside the scope of provincial responsibility. Because provincial governments are not involved in negotiations with or management of native lands, the systems that ensure the safety of drinking water for the majority of Canadians are not applicable to residents of these communities. However, the jurisdictional challenges facing decision-makers are more complicated than optimizing allocation of responsibility. Indeed, chronic water quality problems in native communities are compounded by the challenges of ensuring access to safe water without violating native rights to self-government. We note also that native communities are distinct from the rest of Canada in that basic access to safe drinking water free from pathogenic contamination remains elusive. Here, debate on regulation of trace chemical contaminants is much less important than effective operational procedures for detecting and mitigating risk of short-term illness and death. Management of this type of risk has not been the focus of our analysis so far. Still, our analysis of drinking water in the Canadian context would be incomplete without this discussion.

First Nations communities are the responsibility of the Federal Government and, along with federal lands such as national parks, are not within the jurisdiction of provincial and territorial governments. Therefore, drinking water systems in First Nations communities slip past what is perhaps appropriately termed the 'patchwork' of regulation in place in Canada. Indeed, pathogenic outbreaks and boil-water advisories have become defining characteristics of native communities across the country. In 1995, Health Canada and Indian and Native Affairs Canada[5] (INAC) identified that one quarter of the water systems in First Nations communities 'posed potential health and safety risks to the people they served.' In 2001, INAC found 'a significant risk to the quality or safety of drinking water in three quarters of the systems.' (OAGC 2005a.)

While all parties agree there is a problem, sources differ as to the causes. The OAGC reports that the federal government has spent nearly $2 billion on safe drinking water initiatives in First Nations communities between 1995 and 2003 and attributes continued difficulties to the incomplete scope and intermittent application of federal policies on safe drinking water and inadequacies in the technical expertise available to native communities (OAGC 2005a).

In 2006, INAC announced an action plan to 'ensure that First Nation leaders have access to the tools and resources they need to deliver clean water to their residents' (INAC 2006). Tellingly, the announcement promised access to safe water via new tools and resources rather than a right to safe water. For example, the report does not mention a need for increased accountability on the part of water system operators or a new regulatory or monitoring framework. INAC and Health Canada accepted all of the OAGC's recommendations in the 2005 report with the exception of the institution of a regulatory regime. Instead, they undertook to explore the feasibility of such a program, citing concerns over self-governance on the part of First Nations leaders (OAGC 2005a). Bill S-11, introduced in March 2010, is an effort to close what the sponsors of the Bill refer to as a 'regulatory gap' and allow the governor-in-council to give legal effect to certain types of recommendations related to drinking water on First Nations lands on the recommendation of the Minister of Health (Parliament of Canada 2010). As of April 2012, the Bill is in the hands of the Senate Standing Committee on Aboriginal Peoples, that has been hearing stakeholder comments since February 2011. Thus far, comments indicate that concerns include lack of clarity on funding responsibilities among INAC, Health Canada and Environment Canada, the lack of provisions for funding any regulations that are eventually passed under the Act and a lack of consultation of First Nations peoples (Council of Canadians 2011).

It is beyond our scope to evaluate the quality or extent of the consultations INAC has held. Still, the cost of delaying a regulation likely to improve drinking water quality might exceed the benefits of further consultation. Indeed, the Act serves merely to facilitate individual regulations that may be proposed at a later date and so this

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[5] As of May 2011, this department is known as Aboriginal Affairs and Northern Development

groundwork is essential to move forward with specific regulations that may lead to improved water quality imply further consultation themselves. However, as the Council of Canadians report points out, the Act does not enshrine consultation as a waypoint to more specific regulation.

Beyond Bill S-11, there is some evidence to show that water quality in First Nations communities has improved in recent years. 'High risk' water systems, i.e. those likely to provide unsafe water owing to major deficiencies in several operational aspects, have decreased in number from 193 in 2006 to 49 in 2010 (INAC 2010).

The proportion of First Nations residents considering their water as 'somewhat safe' or 'very safe' increased from 62% in 2007 to 69% in 2009, although this figure remains substantially lower than the 89% of respondents in non-First-Nations communities of similar size (Ekos 2009). Although it is not immediately obvious that this is a significant increase, we see that it is, at least in the statistical sense, in that it is very likely there was a real improvement in respondents' perceptions. The margin of error in the 2009 study is ± 3.7 while in the 2007 study it is ± 2.5, 19 times out of 20 (Ekos 2007). The populations for the 2009 and 2007 studies are 687 and 1 502, respectively. Comparing the outcomes of the two surveys with the two-sample t-test produces a t value of 101.8 which is significant above the 99.9% level[6]:

$$t = \frac{\overline{x_1} - \overline{x_2}}{\sqrt{\frac{\overline{s_1^2}}{n_1} + \frac{\overline{s_2^2}}{n_2}}} = \frac{69 - 62}{\sqrt{\frac{1.89^2}{1502} + \frac{1.27^2}{687}}} = 101.8 \gg t_{0.999,\infty}$$

Still, all actors agree there is a significant disparity between drinking water quality realities and perceptions in native communities, despite the efforts that have been made. Advances in legislation are slow and create anxieties about lack of consultation and funding. We have seen that even with heavy monetary investment, native communities are faced with a shortage of trained personnel and technical ability to implement measures that arguably give 'access' to clean water. Years of effort has led to the development of a regulatory framework that, given the reactions of stakeholders, seems as likely to exacerbate mistrust and create anxieties as it is to bring about improved water quality in the near future. Clearly, the challenges faced in the First Nations context transcend the topics of water quality management we have explored in this paper. Indeed, effective regulation and policy decisions depend on more than science; these decisions must have the trust of all stakeholders. Adding to historical misgivings between First

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[6] Survey results are normal distributions centred at means equal to the point estimate of the outcome (69% in 2009 and 62% in 2007). The standard deviations of both normal distributions (1.27 in 2009 and 1.89 in 2007) produce 95% of the probability density within the margin of error cited by Ekos (± 3.7 in 2009 and ± 2.5 in 2007).

Nations and the Colonial population is the continued lack of transparency, as reflected for example in the Council of Canadians (2011) report, an experience shared by the broader Canadian population.

4.9 Conclusions

Canada stands out from other federations by virtue of the relatively passive role of its federal government and stronger role of provinces and territories. The assignment of responsibility has come about rather naturally in comparison to other countries where assignment to the federal government has occurred via statute, for example with the United States' Safe Drinking Water Act. This decentralized responsibility for drinking water management has produced a highly heterogeneous landscape of drinking water regulation. Public interest in drinking water quality and management surged following several highly reported outbreaks, and several prominent environmental groups have independently and cooperatively weighed in on this issue since the early 2000s. These drinking water reports establish a body of literature claiming inadequate intervention on the part of the federal government and that the resulting framework is under-protective and inequitable, notably with respect to chemical contaminants.

This paper has shown that Canadian political sensitivities to regionalism interact with risk perceptions to energize public opinion in favour of centralizing and strengthening regulatory intervention in drinking water. Concepts of long-term risk imposed by drinking water contaminants and the optimal outlay of public funds are abstract topics that continue to elude experts, to say nothing of a lay public. When dissenting claims of inadequate federal intervention and unequal protection are put in the context of the public's existing uneasiness with the quality of drinking water and a deep cultural aversion to health inequalities, we see to what extent they are likely to reaffirm anxieties and gain currency with an informed public.

The risk perceptions that energize public opposition to unequal drinking water standards may also explain the gap between the foci of academic and popular literature on water governance. As we have seen, the former focuses on developing effective barriers to minimize pathogenic contamination of drinking water, whereas the latter focuses very much on standards for chemical contaminants. This paper has bridged this gap. Through principles of risk analysis and by drawing on the experiences of Canada and the United States, we have shown that drinking water decision-making at the provincial/territorial level is a model more likely to provide cost-efficient, long-term protection of public health. However, we have pointed to two caveats: 1) decentralized decision-making is only likely to be feasible if resource-intensive cost-benefit analysis is centralized and the information made public; and 2) the Canadian public are only likely to support decentralization if provinces and territories are actively allocating public funds to public health initiatives in proportion with the magnitude of risks. In other words, if provinces and territories opt to spend less money on drinking water risk abatement, the Canadian public are likely to demand they use the money saved on addressing a greater public health risk.

Currently, the federal government markets its intervention in water quality management as being directly engaged with the health of Canadians. For example: 'Its mandate and expertise lies in protecting the health of all Canadians by developing the *Guidelines for Canadian Drinking Water Quality* in partnership with the provinces and territories." (Health Canada 2009). Critics are right to question the ability of the federal government to play this role when there is no legal mechanism enabling it to do so; ultimately, it is up to the provinces. In this regard, the federal government might gain legitimacy by embracing its real role as provider of information and expertise to assist more local levels of government that are actually legally responsible for drinking water quality to make rational decisions. It therefore seems that the best use of federal resources would be to ensure access to safe water where it is the responsible level of government and where it has neglected this responsibility, rather than repeating the work of provincial and territorial regulators.

Cost-benefit and contaminant occurrence data are however currently very limited for regulations and guidelines developed in Canada, perhaps because guidelines developed at the federal level do not have the force of law. However, guideline values established at the federal level can be influenced by members of the federal-provincial-territorial subcommittee who want to strike a balance between a health-based standard and cost considerations (O'Connor 2002). Therefore, federal entities intervening in drinking water regulation are subject to the pitfalls of managing drinking water on a national level but are not affected by requirements for transparency and do not facilitate rational decision-making on a more local level. This is perhaps the most serious barrier to efficient allocation of resources in drinking water risk abatement and an invitation to criticisms from commentators on environmental and health policy. On the other hand, an explicit reorientation to data provision and cost-benefit analysis would allow for transparent and cost-effective decision-making as well as enabling the policy research we need to ensure good governance well into the 21st century.

CHAPTER 5

Drinking water regulatory decision-making: Sensitivity to contaminant occurrence and regulatory philosophy in the United States

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Drinking water regulatory decisions made in the United States since the passage of the Safe Drinking Water Act are analyzed in the context of contaminant occurrence patterns, sizes of exposed populations and risks faced by these populations. Two major shifts in the realm of drinking water regulation in the United States that, together, jeopardize the ability of centralized regulation to protect public health are shown to have occurred. The first is a shift in regulatory attention from contaminants relatively widespread and common to many urban areas to those that occur infrequently in a small number of water systems. The second is the entrenchment of a burden on the part of EPA to demonstrate regulatory benefits on a national level for any proposed drinking water regulation. Indeed, recent decisions not to regulate a large number of contaminants were made on the basis that the benefits of regulation would not be sufficiently widespread. It is however demonstrated that the risks posed by these contaminants are as great as or greater than those posed by several contaminants at the time of their regulation by EPA in earlier years. Continued protection of public health depends on an acknowledgement by decision-makers of the challenges posed by the current regulatory framework and the adoption of measures to bring regulatory decision-making into line with the nature of today's contaminants and the public's expectations of the distribution of risks and benefits.

5.1 Introduction

Drinking water regulation emerged in the 20th century as a tool to manage risks posed by source water contamination. In the United States, it has evolved to become both highly dependent on cost-benefit analysis and highly centralized in contrast to, for example, Canada, where responsibility for water quality falls on provinces and territories. Different approaches to drinking water quality management have all enjoyed relative success in delivering consumers water free of pathogens, bacteria and other traditional water contaminants. However, since the entrenchment of centralized regulation in the US, there has been a shift in the nature of contaminants of interest to regulators and the public: falling detection limits now allow us to resolve variation in water quality in terms of a vast number of contaminants at levels on the order of one in a billion. Regional variations in population density, representation of different industries and industrial density, and geophysical characteristics that in some cases exacerbate exposure to environmental contaminants have fragmented our understanding of the drinking water risk geography of the United States and other countries. Regional risk profiles imply diverging priorities for investment of public funds and force regulators to consider not only uneven risk but also unequal value of regulatory benefits.

We are interested in the ability of the centralized regulatory structure in place in the United States to continue to provide cost-effective protection of human health in light of the nature of emerging drinking water contaminants. We show that regulatory attention has in general shifted from contaminants common to many drinking water sources downstream from urban, industrialized areas to contaminants associated to particular industries whose occurrence in water supplies is very exceptional. This change in turn amplifies differences in decision-making outcomes among regulatory structures. In order to pass a regulation on a drinking water contaminant, the US Environmental Protection Agency (EPA) is required to demonstrate that a proposed regulation represents a real opportunity for protection of public health on a national basis, which it has failed to do for 20 out of the 22 new contaminants examined between 2000 and 2011. Meanwhile, many of these same contaminants have been regulated on more local levels in other jurisdictions (see Chapter 4).

Regulatory decisions are justified by cost-benefit analysis and quantified assessments of risk and exposure. However, the interpretation of these analyses is grounded in the socio-political context of the day. The qualitative philosophy governing drinking water regulation is articulated in the 1996 SDWA amendments: for a contaminant to be regulated, it needs to have an adverse effect on the health of persons, occur in public water systems with a frequency and at levels of public health concern and represent a meaningful opportunity for health risk reduction by regulation. Therefore, although there are guidelines for decision-making, they are still relatively subjective and open to interpretation. Before 1996, the regulatory philosophy seems to have been even less explicit. A quantitative analysis of the history of regulatory decision-making reveals the effect of this dynamic interpretation of public health goals.

We examine the drinking water contaminants of interest to US regulators since the early 1960s in terms of their occurrence in drinking water systems and the risks they impose on exposed populations. Comparing these metrics to outcomes of regulatory decision-making, we see the pronounced effects of 1) an increasingly heterogeneous distribution of contaminants and 2) a political climate variably receptive to precautionary regulation. Few states exercise their freedom to regulate drinking water beyond the national standards. Therefore, the changing nature of drinking water risk warrants a new analysis and possibly some reassignment of federal, state and local government responsibilities in providing safe drinking water to Americans. Comparing outcomes of regulatory decision-making to the occurrence and implied risk of drinking water contaminants over time, we see that the underlying regulatory philosophy and interpretation of quality goals overshadows the importance of quantitative analysis in deciding whether to regulate drinking water contaminants. The role of political climate on regulatory decision-making independent of any quantifiable decision-making metrics suggests the need for a more explicit, well-reflected regulatory philosophy that will govern long-term decision-making.

5.2 Background and overview of drinking water regulation decision-making

In the United States, the Safe Drinking Water Act (SDWA) defines the roles of all levels of government in drinking water quality management. At the federal level, EPA is responsible for setting binding regulations, conducting research and monitoring the implementation of the Act. Enforcement is, in turn, primarily the responsibility of state governments (Pontius, 2002). The SDWA was passed in 1974 in response to revelations of widespread water quality problems in early 1970s. The Community Water Supply Study (CWSS) found that drinking water for 2.5 million people served by 397 water systems did not meet the 1962 standards passed by the US Public Health Service (US EPA, 1970). This represented 41% of the water systems and 14% of the population studied. Almost simultaneously, news broke on widespread cancer risks from by-products of water chlorination. On the eve of the passage of the SDWA, synthetic organic chemicals (SOCs) attributable to industrial development were found in the New Orleans drinking water supply, and the newly formed EPA announced a nationwide study to determine the extent of chemical contamination of drinking water (Okun, 2002).

The National Primary Drinking Water Regulations (NPDWR) introduced under the SDWA in 1974 specified maximum levels in finished drinking water largely based on the 1962 Public Health Service standards and findings of the CWSS. In 1976, the Environmental Defense Fund petitioned the EPA for not adequately enforcing the SDWA in part because it had not set standards for any of the myriad SOCs and disinfection by-products identified in drinking water over the previous several years. The subsequent 1978 Court of Appeals ruling instructed EPA to take action on the subject of organics in drinking water. The following year, EPA introduced its trihalomethane rule. The other organics were addressed by the 1986 amendments to the SDWA, mandating that 83 contaminants be regulated by 1989. This included a number of contaminants that had already been regulated under the 1974 regulation, with interim deadlines in 1987 and 1988. Each contaminant was to be regulated as close as ‘feasible’ to the maximum contaminant level goal (MCLG), a quantity that represents the highest concentration of a contaminant in drinking water believed with confidence to be safe. This condition forced the revision of some regulations from the initial criteria promulgated in 1974. For many contaminants, particularly carcinogenic contaminants that are assumed to carry some level of risk at any dose, the MCLG is zero. Determining how close to zero a contaminant can feasibly be regulated therefore requires balancing costs of compliance with the associated public health benefits.

By 1992, EPA had regulated 82 of the 83 contaminants required by the 1986 SDWA Amendments[7]. Although EPA had the responsibility of deciding what MCL was ‘feasible’ for the prescribed contaminants, it did not identify *which* contaminants to

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[7] Sulfate was never regulated

regulate out of the hundreds of thousands of possibilities. Its only discretion was the latitude to choose up to seven of the 83 mandated contaminants to substitute with contaminants that in EPA's judgement represented greater opportunity for public health protection[8] (US EPA, 1989). This top-down regulatory process consisted essentially of lawmakers decreeing regulatory priorities and obligating EPA to set health-protective values. However, the 1986 Amendments did require EPA to itself identify regulatory priorities following regulation of the first 83 contaminants: 25 new drinking water contaminants were to be identified and regulated every three years starting in 1991. This requirement was never met: no new contaminants were regulated between 1993 and 1998, when the 1996 SDWA came into effect and established the framework in place today. The 1986 Amendments were passed against the backdrop of calls to reduce the burdensomeness of regulation, as exemplified by President Reagan's 1981 executive order no. 12291, requiring that the benefits of new regulation 'outweigh' the costs. The difficulty of meeting this condition contextualizes EPA's inability to identify and regulate new drinking water contaminants beyond the mandated 83. President Clinton's executive order no. 12866 decreased the burden of proof such that the benefits only had to 'justify' the costs (Morgenstern, 1994).

Since 1998, EPA has been required simply to evaluate five contaminants for regulation every five years, with no regulation quota. EPA now compiles contaminant candidate lists (CCL) to track opportunities for drinking water regulation and has since 1998 made 20 regulatory determinations. None of the CCL contaminants were found to represent meaningful opportunities for public health protection, because they are too rare. Beyond these 20 contaminants, EPA regulated uranium and has decided to regulate perchlorate and made revisions to existing standards for radiological contaminants and arsenic. Therefore, since 1998, EPA has decided against regulation for 20 out of the 22 new contaminants it has examined as candidates. These determinations are made on the basis of occurrence and toxicological data and test the applicability of the three statutory requirements of a proposed drinking water regulation under the 1996 Amendments, described in Section 1.. The large majority of the contaminants from CCL1 and CCL2 that EPA did not regulate failed to meet the condition of sufficient occurrence in public water systems.

States have the freedom to pass drinking water regulations as long as they are not less stringent than federal standards, but as of 2011, only three have done so. Residents of California, New Hampshire, New Jersey and New York benefit from state-level legally binding drinking water quality standards, with more localized analysis comparing drinking water risks to abatement costs in the context of a more localized portrait of risk reduction priorities. The diverse conditions of the remaining 243 million Americans are

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[8] Removed : aluminium, dibromomethane, molybdenum, silver, sodium, vanadium, zinc. Added: aldicarb sulfone, aldicarb sulfoxide, ethylbenzene, heptachlor, heptachlor epoxide, styrene, nitrite

aggregated into the cost-benefit models of the EPA to monitor and forecast the relationship between costs and benefits of existing and potential drinking water regulations. Indeed, cost-benefit analysis plays a crucial role in drinking water regulation, determining the level of intervention where decision-makers are unconstrained by detection ability and where dose-response models imply additional risk at any dose greater than zero (Calder and Schmitt, 2010). Numerical standards are only one part of drinking water quality management and would be ineffective without a framework ensuring adequate monitoring, water supply operator training, treatment and so on. Still, the numerical standards determine the end goals of these other water management strategies, define how quality is measured and essentially govern decisions on source water selection and treatment requirements. They are perhaps the single most important element of drinking water quality protection.

5.3 **Occurrence, distribution and regulatory outcome of drinking water contaminants over time**

We illustrate the increased heterogeneity and localization of drinking water contaminants of interest to regulators since the 1980s, as compared to the regulatory priorities of the 1960s and 1970s. A comparison of regulatory determinations in the 1980s to the 2000s furthermore shows that different outcomes are possible for contaminants with similar occurrence profiles. We compare regulatory decision-making outcomes on the basis of risk profiles in the next section. We show that as attention shifted from contaminants widespread in water systems to contaminants that affect discrete populations, regulation of these new contaminants was possible due to a political culture open to precautionary regulation. Political culture began to change after the 1986 amendment, demonstrated by an aversion to government intervention as a solution to environmental problems. EPA therefore faced increased difficulty in justifying regulations of rare contaminants that would add a burden of compliance nationwide.

The CWSS provided the bulk of the data on contaminant occurrence in public drinking water systems used to develop the NPDWR of 1975. The National Organics Reconnaissance Survey (NORS) and the National Organics Monitoring Survey (NOMS) of the mid-1970s were undertaken in order to investigate the extent of disinfection by-product and synthetic organic chemical contamination in drinking water. The trihalomethanes rule of 1979 and the SDWA amendments were informed by these studies. The CWSS aimed to characterize drinking water quality in all sizes of water system, in urban and suburban areas in all regions of the continental US in terms of the 1962 PHS water quality criteria. The 1962 standards made a distinction between standards that indicate that other drinking water sources should be used, where available, and standards that indicate a given supply should be rejected (US PHS, 1962). EPA referred to these as recommended and mandatory standards, respectively (US EPA, 1970). System noncompliance and size distribution data from the CWSS were combined to analyze the extent of water quality problems in the US at the beginning of the 1970s. The CWSS provided noncompliance data in terms of percentages of systems in the study but generally not in terms of populations affected. The CWSS did however identify how many water systems fell into different ranges of populations served. We used the ranges to create low and midpoint assumptions about the sizes of the water systems in each range that pass or fail mandatory or suggested standards. Calculations using high assumptions were excluded because the midpoint estimate is already an adequate upper bound. This is for example the case for total exceedences (mandatory or recommended, all system sizes combined): EPA reported 2.5 million Americans served by drinking water not meeting either recommended or mandatory standards, compared to our midpoint estimate of 5.2 million and our low estimate of 1.6 million. About 30% of Americans were therefore served by water not meeting the 1962 standards (recommended or mandatory) with higher proportions in smaller water systems, particularly in the case of mandatory standards. For example, while only 1 to 2% of the total survey population was exposed to drinking water out of compliance with the *mandatory* limits, this figure jumps to 5 to 10% when we consider only water systems serving 25 000 people or less, and up to 24% for water systems serving 500 people or less. This cursory inspection demonstrates the severe water quality issues in pre-SDWA America. Table 5.1 summarizes this analysis.

CWSS data on violation of standards and the sizes of exposed populations were compared to the values of the subsequently enacted NPDWR. Generally, the NPDWR addressed those contaminants that the CWSS showed to be adversely affecting water quality on a national scale. The CWSS in turn included contaminants targeted by the 24 recommended or mandatory 1962 PHS standards, with several exceptions. The CWSS tested water for 25 contaminants including boron and turbidity and excluding phenols, differing in that respect from the 1962 PHS standards. Of the 25 contaminants tested within the CWSS, there were detections beyond either the recommended or mandatory standards in 20, with 15 having detections exceeding standards in more than 1% of the public water systems examined. In response, the NPDWR targeted 16 contaminants, seven of which (four pesticides[9], two herbicides[10] and mercury) had not been evaluated by the CWSS, and one of which, silver, *had* been evaluated by the CWSS but found not to exceed the PHS standards. The remaining eight had been tested by the CWSS and were found in levels exceeding the PHS standards, although four of these affected less than one percent of all PWS tested. Finally, ten contaminants detected by the CWSS in levels exceeding the recommended 1962 PHS standards (but none exceeding the mandatory standards) were not regulated by the NPDWR. This analysis is summarized in Table 5.2.

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[9] Endrin, lindane, methoxychlor and toxaphene

[10] 2,4-D and 2,4,5-TP (Silvex)

This comparison of the results from the CWSS with the ensuing NPDWR serves mainly as an illustration of the widespread benefits of the earliest drinking water regulation. However, the regulation of four pesticides and two herbicides under the NPDWR is noteworthy in another sense: EPA does not seem to have considered its occurrence in drinking water supplies when developing the regulation. Beyond the lack of inclusion in the CWSS or discussion of occurrence in the supporting material for the interim NPDWR (US EPA, 1976), studies available at the time of regulation indicate that these contaminants occur rarely and at orders of magnitude below the applicable health levels in drinking water and surface water (for toxaphene, see US EPA, 1971a and for 2,4-D and 2,4,5-TP see US EPA, 1971b). This is antithetical to EPA's current strategy for priority-setting that focuses strongly on contaminant occurrence in water systems and has been the grounds for almost all the negative regulatory determinations made since 2000.

The National Organics Reconnaissance Survey (NORS) was carried out in 1975 to determine the extent of occurrence of trihalomethanes (THM)[11], a class of suspected by-products of the drinking water chlorination process previously detected in samples of finished drinking water. This study also examined two chlorinated chemicals[12] previously detected in finished drinking water but not necessarily linked to water disinfection (US EPA, 1975a). Table 5.3 displays the results from the NORS study and reflects the finding that they were generally widespread. The water systems contributing data to the study were chosen to reflect a wide range of raw water sources and treatment practices but were not analyzed in terms of populations served in order to quantify population exposure. The statistics in Table 5.3 therefore refer to the distribution of results, with each water system out of the total of 80 providing one sample. The NORS report handles total trihalomethanes in terms of moles per litre, that is, adjusted to express total concentration in terms of total number of molecules. Because subsequent regulation and analysis has tended to be in terms of a simple sum of the concentrations without regard to differences in molecular weights, Table 5.3 displays total THMs in terms of µg/L, calculated manually from the raw data in the NORS report. Where individual water systems reported any of the individual THMs as 'not found', a concentration for that THM equal to the average detection limit displayed in Table 5.5 was used to calculate total THM for that data point, unless the water system gave information about the particular detection limit (e.g. 'not found in levels above 1 µg/L'), in which case that detection limit was used. Alternatively, total THMs could have been calculated using a concentration of 0 wherever 'not found' was reported without a significant effect on the result.

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[11] Chloroform (trichloromethane), bromodichloromethane, dibromochloromethane and bromoform (tribromomethane)

[12] 1.2-dichloromethane and carbon tetrachloride

The original THM rule passed in 1979 set the maximum concentration of total THMs at 100 µg/L. This was revised in 1998 to 80 µg/L at the same time as other disinfection by-products were regulated, notably the group of haloacetic acids (US EPA, 2011). It is not clear to what extent the health effects of THMs were understood by regulators at the time of regulation in 1979. As of 1975, EPA considered the risks to human health real but unquantifiable (US EPA, 1975b). In any case, the original regulation of 100 µg/L corresponds to the 80th percentile of the water supply systems sampled in NORS. The National Organics Monitoring Survey (NOMS) conducted a more extensive analysis of total THMs, among other compounds (US EPA, 1977). From this data set, the 1979 regulation of 80 µg/L corresponds to the 70th percentile of water supply systems sampled. Regardless of our confidence in the underlying toxicological basis for this level of regulation, it is clear that the 1979 rule affected a substantial portion of Americans served by chlorinating public water systems.

Data from cities reporting to NOMS were harder to summarize on a comparative basis than those from NORS: reporting cities returned between one and four samples for a given contaminant, each with different handling methods. For a given contaminant and a given reporting city, there were often detections in some samples and non-detections in others with most results of most parameters being non-detects at detection limits relatively close to the median value of detections. In this case, our treatment of non-detects has a more significant bearing on the representation of the data set. We have therefore excluded the non-detects from the data altogether and the other results for a given contaminant were averaged to represent the contaminant concentration for a given reporting city. The exception to this was one result for 1,2,4-dichlorobenzene, where the average of the two detections would have been equal to 50.2% of the higher detection, thus skewing the result downward. Although this preserved higher value is in effect the maximum value out of all reporting cities for this contaminant, as displayed in Table 5.3, there is no impact on subsequent analysis because of the higher value of the ensuing regulation. Averaging the results for the other reporting cities did not otherwise mask high detections. The percentile data for the population of detections is presented alongside the size of this population for each contaminant, out of a total of 113 systems tested, to demonstrate how many points were censored to the left of the detection limit.

Table 5.3 compares the NOMS findings to those from NORS and to the regulations that resulted from the 1986 SDWA amendments as discussed in Section 2. Of the 83 contaminants set to be regulated under the 1986 amendments, 13 had been studied in NORS or NOMS[13]. Nine of these 13 contaminants were regulated at a level greater than the maximum detection in either of the studies[14], suggesting that the utility of these

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[13] Counting total but not individual trihalomethanes

[14] Including 1,2-dichloroethane whose maximum detection in NOMS and carbon tetrachloride whose maximum detection in NORS were less than their eventual regulatory levels.

regulations is constrained to avoiding exposure of consumers to water with levels higher than the MCLs adopted, that is, pre-emptive regulation. Thirteen contaminants between the NORS and NOMS were found in at least ten public water systems nationwide, eight of which were regulated under the 1986 Amendments. The other five contaminants regulated under the Amendments were found in fewer than ten water systems nationwide, notably vinyl chloride and ortho-dichlorobenzene, with two and three detections respectively in NOMS. These two examples are also among the nine that were regulated to levels higher than the maximum level detected. Only three contaminants regulated under the 1986 Amendments were at least somewhat widespread (detected in at least ten water systems) and present at levels higher than the ensuing respective regulations: total THMs, 1,1,2-trichloroethylene and carbon tetrachloride.

Notably, two of the seven contaminants EPA opted out of regulating to the lowest level feasible under the 1986 SDWA Amendments (silver, which had already been regulated under the NPDWR, and zinc) had occurrence data from either the CWSS, NORS or NOMS. However, none of the seven contaminants substituted for regulation had occurrence data either. This is a powerful indicator that real occurrence in public water systems was not a strong factor in EPA's decision-making at the time of implementation of the 1986 Amendments.

The 1998 Stage 1 Disinfectants and By-products Rule (DBPR) adjusted the total THM regulation from 100 to 80 μg/L and set standards for five haloacetic acids (HAA5)[15] and specific disinfection residuals. The 1998 Enhanced Surface Water Treatment Rule (ESWTR) addressed microbiological agents cryptosporidium and giardia in addition to revising the turbidity standard. The 2000 radionuclides rule revised the standards for gross alpha and beta, radium 226 and 228 and established a uranium standard. The 2001 arsenic rule revised the existing standard based on revised toxicological information and feasibility of implementing a lower level (US EPA, 2009).

The occurrence data for these last examples are hard to analyze in a fashion symmetrical to what we have presented so far. The principle data set for occurrence of contaminants and disinfection residuals addressed by the Stage 1 DBPR came from the survey carried out under the Information Collection Rule (ICR). The data are presented as numbers of treatment plants presenting detections within certain ranges (e.g. 13 groundwater plants with detections of total THM between 20 and 30 μg/L in the period between July and September 1997). Because the individual data points are not specifically known and can only be ordered by range, and because of some censoring issues (the highest category is >130 μg/L), percentiles can only be expressed in terms of ranges. A similar approach was adopted for chlorite and HAA5. The data for bromate could not be meaningfully analyzed in a similar way because the majority of the data points were censored in the > 30 μg/L

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[15] Dichloroacetic acid, trichloroacetic acid, chloroacetic acid, bromoacetic acid and dibromoacetic acid

range. We can only say therefore that the maximum, 95th percentile and median values are greater than 30 μg/L. All occurrence data for the contaminants regulated between 1998 and 2001 plus perchlorate are summarized in Table 5.4, with the exception of chloramine and chlorine dioxide. These latter two compounds are disinfection chemicals, and their concentrations are a product of water treatment practice. They differ in this way from other contaminants in that there is no characteristic of source water quality that contributes to their occurrence in finished water.

The principal radionuclide occurrence data used for drinking water regulatory decision-making in the 1990s and 2000s came from the National Inorganic and Radionuclides Survey (NIRS) on drinking water and a subsequent study on radionuclides in groundwater carried out by the United States Geological Survey (USGS). (65 FR 21576). These studies are in turn discussed in the technical support document for the 2000 radionuclides rule (US EPA, 2000). Establishing percentile ranks for radionuclides is difficult because the bulk of results for most contaminants are below the reporting threshold. Table 5.4 shows, essentially, that the regulations are high compared to the bulk of results. Still, the technical support document indicates that about 10 000 water systems (roughly half of them serving fewer than 500 persons) out of a total of 53 000 exceed the MCLs adopted for one or more radionuclides. Therefore, radionuclide occurrence with respect to the ensuing regulatory values was rather more widespread than the bulk of contaminants regulated under the 1986 SDWA Amendments. Similarly, the 2001 revised arsenic regulation of 10 μg/L falls below the 95th percentile of occurrence, and the updated disinfection byproducts rule had implications for relatively large populations. All our estimates for arsenic occurrence after the 1990s are based on EPA's technical fact sheet (EPA, 2001) and occurrence summary document, drawing together data from SDWA compliance monitoring activities, the USGS database, the NIRS and from the Safe Drinking Water Information System (EPA, 2000).

Data for the 20 contaminants EPA assessed and decided not to regulate within the CCL since 2001 were analyzed and compiled in Table 5.5. These are the first contaminants chosen from the first two contaminant candidate lists (CCL1 and CCL2) to have undergone regulatory decision-making. Also included is perchlorate, which EPA has decided to regulate after extensive analysis and consultation. For context, the health reference level (HRL) calculated by EPA for each contaminant is provided. In the case of noncarcinogenic contaminants, this level is based on the level of no observed adverse effect (NOAEL), or the lowest observed adverse effect level (LOAEL). For carcinogenic contaminants, the HRL generally corresponds to the one-in-a-million (10^{-6}) additional cancer risk level. From Table 5.5, we see that five of the 19 contaminants were never found in levels exceeding the HRL. However, we also see that significant populations across the US are exposed to the contaminants considered for regulation at levels of potential public health concern. The majority of these are not especially localized. Exceptions are DCPA, detected only in Michigan at levels above the HRL; DDE, detected only in Alabama at levels above the minimum reporting level (MRL) and hence the HRL, as in this instance HRL > MRL; 2,4-DNT detected only in Tennessee at levels above the MRL and hence the HRL; and hexachlorobutadiene, detected at significant levels in Florida and also but to a lesser extent in Alabama, Arizona, New Jersey, Ohio and Tennessee. It is striking that the 'low' levels of occurrence (relative to the total US population) are cited as reasons precluding regulatory action when so many regulations in the past have been passed without evidence of occurrence at levels of concern.

The contaminants examined for regulation from CCL1 and CCL2 plus perchlorate have relatively detailed occurrence information, collected within the Unregulated Contaminant Monitoring Program Rounds 1 and 2 (UCM Round 1 and UCM Round 2, 1988-1997) and within the first phase of the UCM program as revised by the 1996 amendments to the SDWA (UCMR 1, 2001-2005). The notation used to describe these three data sets follows that of EPA (e.g. 2011) We see that most of these contaminants are expected to affect populations on the orders of 100 000 to 1 000 000 at levels of concern. One exception is perchlorate, for which EPA has developed a range of HRLs, from 1 to 47 μg/L, based on its effects on people at different life stages. Considering an HRL of 1 μg/L, the size of the exposed population is an order of magnitude greater than that of the high end of the other contaminants (> 16.6 million). However, this is the total population, not the size of the relevant subpopulation the HRL targets. Still, even at the maximum HRL of 47 μg/L, corresponding to the least sensitive life stage, the exposed population is still between 110 000 and 400 000.

However, these populations are connected to a relatively small number of water systems. In most cases, the 99th percentile of detections is below the reporting limit, and a meaningful statistical analysis can only be done on the subpopulation of detections (US EPA, 2002 and 2003a to 2003l). The occurrence profiles of the post-2000 CCL contaminants are therefore similar to those regulated under the 1986 Amendments, with low doses of perchlorate being consistently more present than the others. The most striking difference is the relative availability of occurrence data for the CCL contaminants. Evaluating occurrence of CCL contaminants with respect to their respective health reference levels is a useful way to measure how widespread contamination is at levels of concern. It would therefore be useful to make this comparison for contaminants regulated in the 1970s and 1980s, but this would not be a fair comparison as this concept surfaced in analyses on drinking water regulatory decision-making only in the 1990s.Although the number of individuals exposed to these contaminants of concern is not negligible, the contaminants fail to meet the subjective regulatory criterion of being present sufficiently often.

Drinking water regulations passed before the 1980s were in general widely protective of public health and relevant to large numbers of water systems, with regulatory limits in general lower than contaminant concentrations commonly encountered. However, we have also shown that widespread contaminant occurrence in this period was not a necessary condition to regulation, with several contaminants regulated without occurrence data or even where available data suggested that occurrence was rare. Contaminants regulated under 1986 Amendments were primarily organic and decidedly rarer and present in lower concentrations than those contaminants previously of interest. Occurrence of these contaminants was mostly not known, and for most of those contaminants whose occurrence was known, the regulatory value was higher than the maximum concentration detected. Indeed, regulation of this new class of contaminants was almost purely pre-emptive. For the most part, this pattern of contaminant occurrence has continued to the present day. Regulation has however generally not occurred, signalling a shift away from pre-emptive action on the part of regulators. Figure 5.1 plots the shift in contaminant occurrence characteristics and the shift away from precautionary regulation on top of a drinking water regulatory timeline.

5.4 Risk-based analysis of regulatory decisions

Carcinogenic contaminants provide an opportunity for a more quantified analysis of the changing drinking water regulatory landscape. Although the occurrence of noncarcinogenic contaminants can be assessed with respect to their HRLs, their varying health effects and non-quantified levels of toxicity make comparisons difficult. On the other hand, carcinogenic contaminants are generally modelled by EPA as having a linear dose-response model with a quantified slope and calculable additional cancer risk levels at different doses. This makes comparison, for example for the purpose of measuring the risk posed by different contaminants, relatively easy. EPA uses weight-of-evidence (WOE) rankings to evaluate the carcinogenic potential of different substances. We use a binary approach and divide contaminants broadly between ‘carcinogenic’ (meaning EPA has assigned a carcinogenic dose-response relationship) and ‘non-carcinogenic’, including those contaminants that are not classifiable with respect to carcinogenicity on account of insufficient evidence.

We use this approach to estimate the number of additional cancers expected on the basis of occurrence values of different carcinogenic contaminants, both previously regulated and assessed for potential regulation by EPA. This provides an objective measure of how regulation has addressed cancer risks over time. Retrospective assessment of risk countered by regulations pre-1980 is however problematic, because the tools developed for this purpose did not exist and were not used by regulators as they are today. The risk analysis tools used by regulators today emerged with National Academy of Science’s ‘Red Book’ (NRC, 1983), and the toxicological information used to derive dose-response models for different contaminants has accumulated to the present day. Calculating risk information for contaminants regulated before the 2000s can only be done on the understanding that this information was not available to regulators at that time.

Occurrence data for carcinogenic contaminants regulated since 1960 and contaminants studied within the CCL process were combined with EPA dose-response models to determine risk levels associated with higher exposures of contaminants over the years. This analysis is summarized in Table 5.6. As discussed, almost all samples for most contaminants analyzed within the CCL process returned results of non-detect. To estimate the 95th percentile exposures and populations for the CCL contaminants, we have assumed overlapping triangular distributions plotted between the modal value of the detection limit, the maximum occurrence value and centred on the median, values provided in the regulatory support literature (US EPA 2002 and 2003a to 2003l). The 95th percentile concentration thus calculated is assumed to affect 5% of the population. This is likely to be somewhat of an overestimate, given that water quality problems affect small systems disproportionately. Exposure estimates for pre-CCL contaminants are those described in the previous section with the exception of arsenic, for which one additional estimate is made: the population above the regulatory level at the time of regulation for the original NPDWR of 0.05 μg/L. This estimate is expressed as a range and was made on the same basis as the average and low population estimates from data on system violations described in the CWSS and summarized in Table 5.1. The low and average estimates were calculated assuming a 1% exceedance rate for arsenic. However, the value was reported as < 1%. Still, we saw in the previous section that the low end of the range can be a significant underestimate, so the lower end of an estimate for 1% might be a good approximation for < 1%.

As a general guideline, EPA generally considers lifetime risks on the order of one in a million to be small or *de minimis*, and risks on the order of one in ten thousand to be significant. Along these lines, we find two contaminants that present large risks to the more highly exposed population: aldrin and dieldrin, with additional lifetime risks on the order of 1.4 in 1000. Beyond these two contaminants, 1,1,2-2-tetrachloroethane may represent a substantial risk to some populations, with a 95th percentile value between 3.3 in 1000 and 6.7 in 1 000 000, according to the UCM Round 1 and Round 2 data, respectively. Populations exposed to risks at levels equal or greater to these are estimated at 40 000 and 230 000 respectively. 1,3-dichloropropene presents a 95th percentile risk between 7.6 in 100 000 and 4.3 in 1 000 000 according to Round 2 and Round 1 data respectively. Populations exposed to risks equal or greater to this are estimated at 58 550 and 100 800, respectively.

Notably, aldrin and dieldrin were not regulated because of their rare occurrence and because occurrence is expected to decline in future years as these compounds are no longer in use. The risk levels cited above assume *lifetime* exposure, and so occurrence over a smaller timeframe will reduce the risk accordingly. This would be the case if exposure to aldrin and dieldrin declines over a period of less than a generation. It is unclear whether this is indeed happening; usage was discontinued in 1989 (ASTDR, 2002) and significant levels remain in the drinking water. Regardless of future trends in drinking water occurrence, this decision contrasts to earlier policies of regulating preemptively where there was an opportunity to protect public health.

The large majority of carcinogenic drinking water contaminants currently regulated under the SDWA were regulated under the 1986 Amendments. As we have seen, occurrence data were not available at that time for most of these contaminants. Still, these contaminants can be evaluated in terms of the additional cancer risk posed at the level of regulation as an indication of the maximum risk permitted attributable to each contaminant. We estimate the size of the population benefiting from regulation of arsenic at the 0.05 μg/L level in 1974 to be in the 5 to 37 million range. EPA's estimate for the population size benefiting from regulation at the 0.01 μg/L level in 2001 is 11.7 million (US EPA, 2000). Populations benefiting from regulation of bromate and carbon tetrachloride were not estimated. Notably, however, for all regulated carcinogenic contaminants on which occurrence data *were* available at the time of regulation, with the exception of arsenic, bromate and carbon tetrachloride, we see that no populations were found to be exposed to levels at or above the ensuing regulatory level. That is, with those three exceptions noted, all regulated carcinogenic contaminants for which occurrence data were available at the time of regulation were regulated with a purely precautionary approach. The risk levels corresponding to the resulting regulatory level for these contaminants regulated on a precautionary basis are on the 2.2 in 1 000 000 to 8.4 in 100 000 range.

Therefore, the populations exposed to carcinogenic risks resulting from CCL contaminants at the time of their regulatory evaluation are in general larger than the populations that were facing comparable or smaller carcinogenic risks at the time of their regulation. It is however worth restating that the framework and dose-response data for this quantitative analysis was not in most cases available at the time of regulation of regulation of pre-CCL contaminants.

5.5 Conclusions

We have conducted a comprehensive analysis of occurrence patterns of the drinking water contaminants of interest to United States regulators from the 1960s to the 2010s. We have identified a shift in regulatory attention from contaminants widespread in drinking water supplies up to the 1970s towards contaminants that occur generally infrequently and at trace levels. We have also identified a decreasing tendency to regulate drinking water contaminants preemptively and increased burden to demonstrate nation-wide benefits from proposed drinking water regulatory intervention. The combination of the shift in the nature of drinking water contaminants of interest with the decreased openness to precautionary regulation appears to have left regulators deadlocked. An analysis of regulation in terms of risks faced by exposed populations, and of the size of these populations, confirms that the opportunities to protect public health afforded by prospective regulation of today's contaminants are comparable to those previously addressed by regulatory intervention. Studying the exposure profiles of drinking water contaminants which emerged in the past 40 years and the recent decision-making history of the EPA, it seems unlikely that many contaminants will be identified that are present sufficiently often and in sufficiently high concentrations to warrant regulation. The case of perchlorate demonstrates that regulation in this framework is not impossible, but contrasted with the outcomes of the 20 contaminants from CCL1 and CCl2, regulation nevertheless seems unlikely. Moreover, a final regulatory value for perchlorate has yet to be issued more than a decade after analytical chemistry techniques first enabled detection below a provisional reference dose, suggesting that the road to regulation for new contaminants may be difficult. (US EPA 2002b)

The centralized nature of drinking water regulation in the United States does indeed imply that the costs and benefits of any proposed regulation should be shared somewhat equitably across the country. Significantly, the regulatory analysis for the CCL contaminants essentially stopped upon determination that each was not present with sufficient frequency across the country, presumably leaving the regulatory burden to more local levels of government. Individual states are free to enact regulations that may be appropriate on more local levels. However, the fact that only a handful of states have regulations beyond those required by the EPA implies that this is not happening in practice. This is possibly attributable to the expense of regulatory analysis, lack of technical expertise at the state level or public aversion to the state duplicating regulatory activity already covered at the federal level. A striking example of this last point is New Jersey Governor Chris Christie's executive order number 2, given in 2010, that requires state agencies to "reduce or eliminate areas of regulation where federal regulation now adequately regulates the subject matter".

In the interest of public health protection, EPA could facilitate decision-making on the part of states having large populations that would benefit from regulation. We note that EPA essentially stops the regulatory analysis for a CCL contaminant once it has demonstrated that the contaminant does not occur at sufficiently high levels or with sufficient frequency to warrant regulation by EPA. However, it remains silent on the costs and benefits of regulation 1) by EPA itself or 2) on a more local level. Indeed, this analysis appears to fall well outside the framework in which regulatory evaluations are currently made. More generally, there is a need for a comprehensive reevaluation of how drinking water regulatory responsibility is divided among levels of government in order to develop a regulatory strategy compatible with the nature of today's contaminants and protective of public health at a reasonable cost.

CHAPTER 6

Summary of research contributions

The work presented in this thesis has grounded the candidate in the foundations of an integrated approach to water resources management and represents novel contributions to the extant sub-disciplines of drinking water and wastewater management and decision-making.

Chapter 1 uses CFD techniques to generate new understanding of the hydrodynamic characteristics and optimal operating conditions of a new type of wastewater treatment technology.

Chapter 2 provides a framework for integrating newly accessible techniques in environmental fluid mechanics to wastewater management decision-making. It demonstrates the applicability of risk analysis techniques to structure CFD modelling research and outlines new paths of inquiry in ecological and economic impact valuation that would allow for this framework to become fully operational.

Chapter 3 investigates a recurring claim that improved detection ability leads to regulations stricter than justified by available toxicological data. It outlines how the regulatory apparatus responds to emerging contaminants as a first step to developing a predictive regulatory structure.

Chapter 4 explores centralized and decentralized approaches to managing drinking water in Canada from the perspective of theoretical risk management principles, risk perception, practical barriers such as the costs of decision-making and quality outcomes of disadvantaged populations. It finds that the popularly demanded greater centralization of regulatory responsibility is not likely to have intended outcomes of improved water quality, with the possible exception of native communities. It also finds that greater centralization will also make decision-making more complicated and likely slow responsiveness to emerging contaminants.

Chapter 5 puts the history of drinking water decision-making in the US in the context of risk abatement to see if decisions to regulate and to not regulate have been consistent. It finds that as drinking water contaminants become rarer and the exposure landscape becomes more fragmented, contaminants tend not to be regulated, even where the risks are equal to or exceed risks addressed by previous regulations. This further points to the advantages of greater decentralization in drinking water decision-making.

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FIGURES

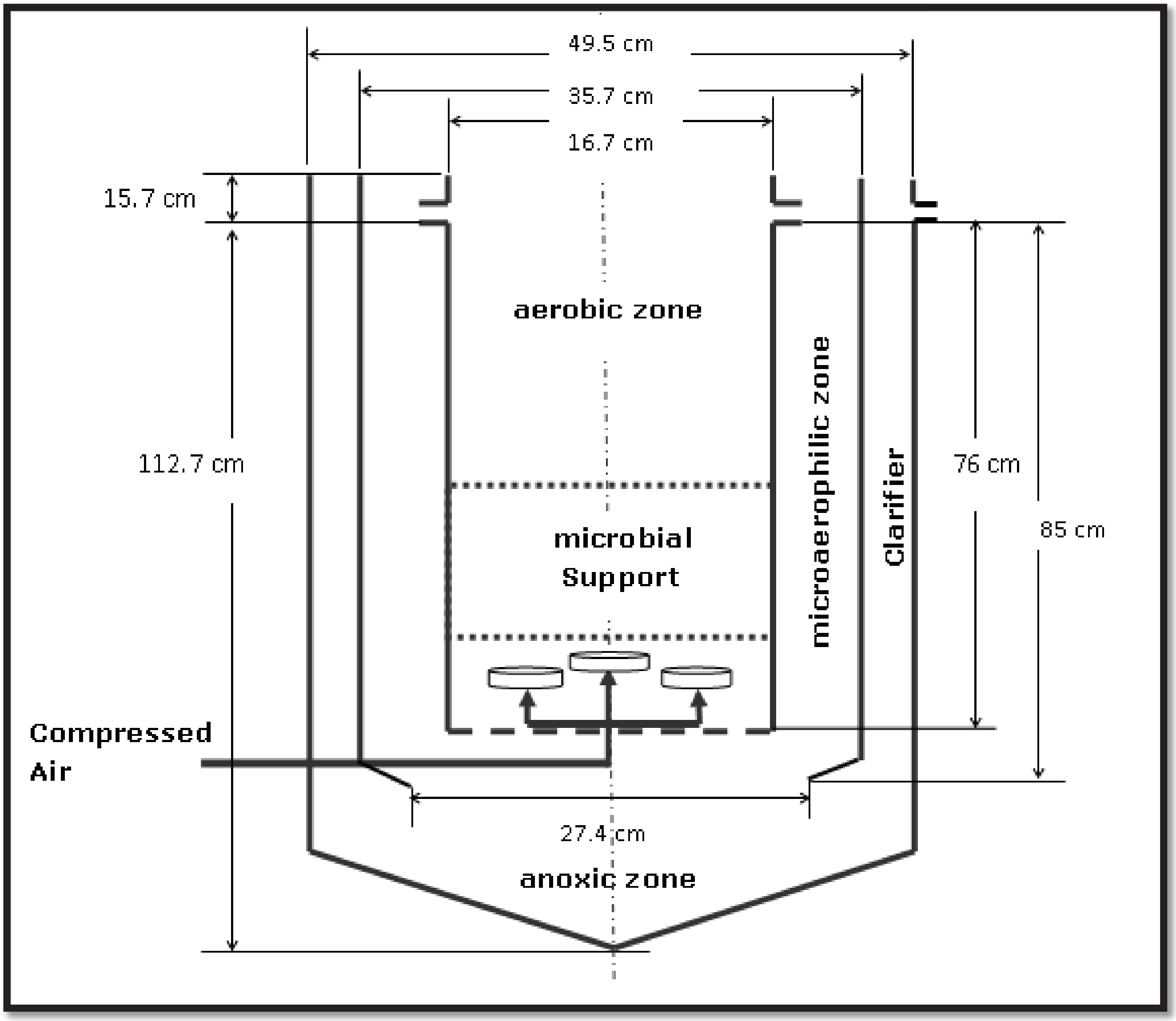


Figure 1.1: Schematic of laboratory-scale BioCAST reactor

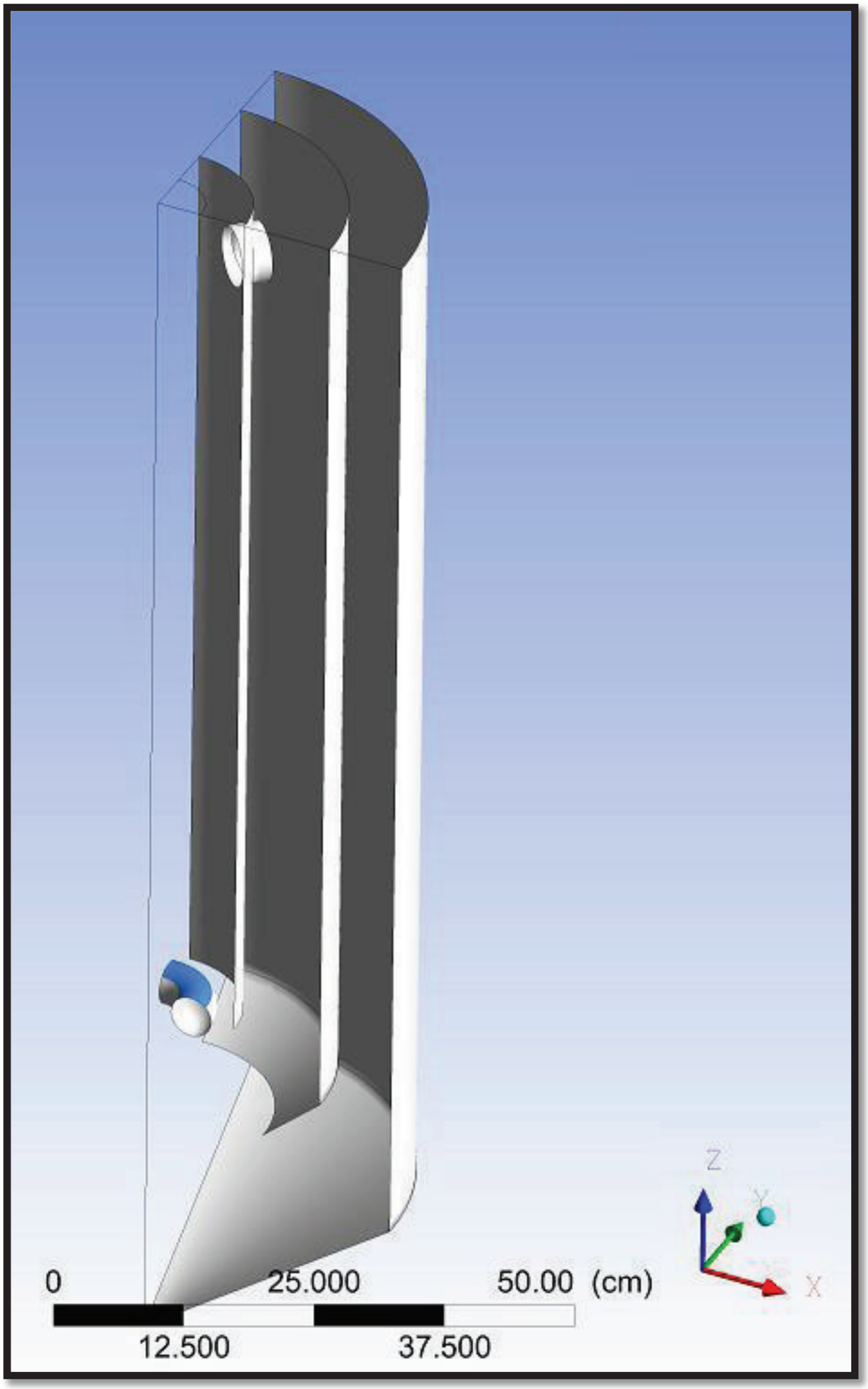


Figure 1.2: CFD model geometry visualization

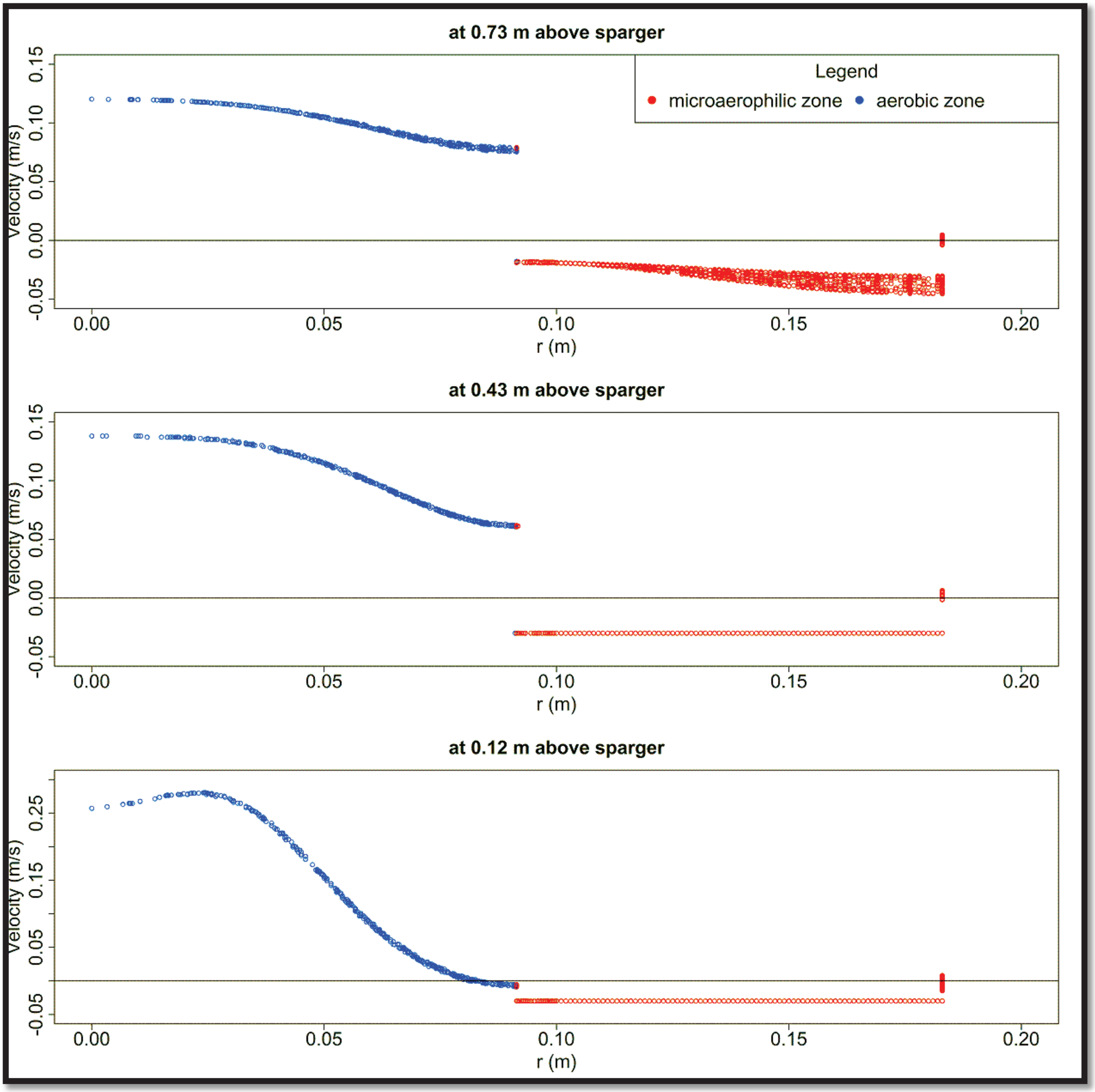


Figure 1.3: Vertical (Z-direction) velocity at three elevations in the bioreactor modelled in Run 10

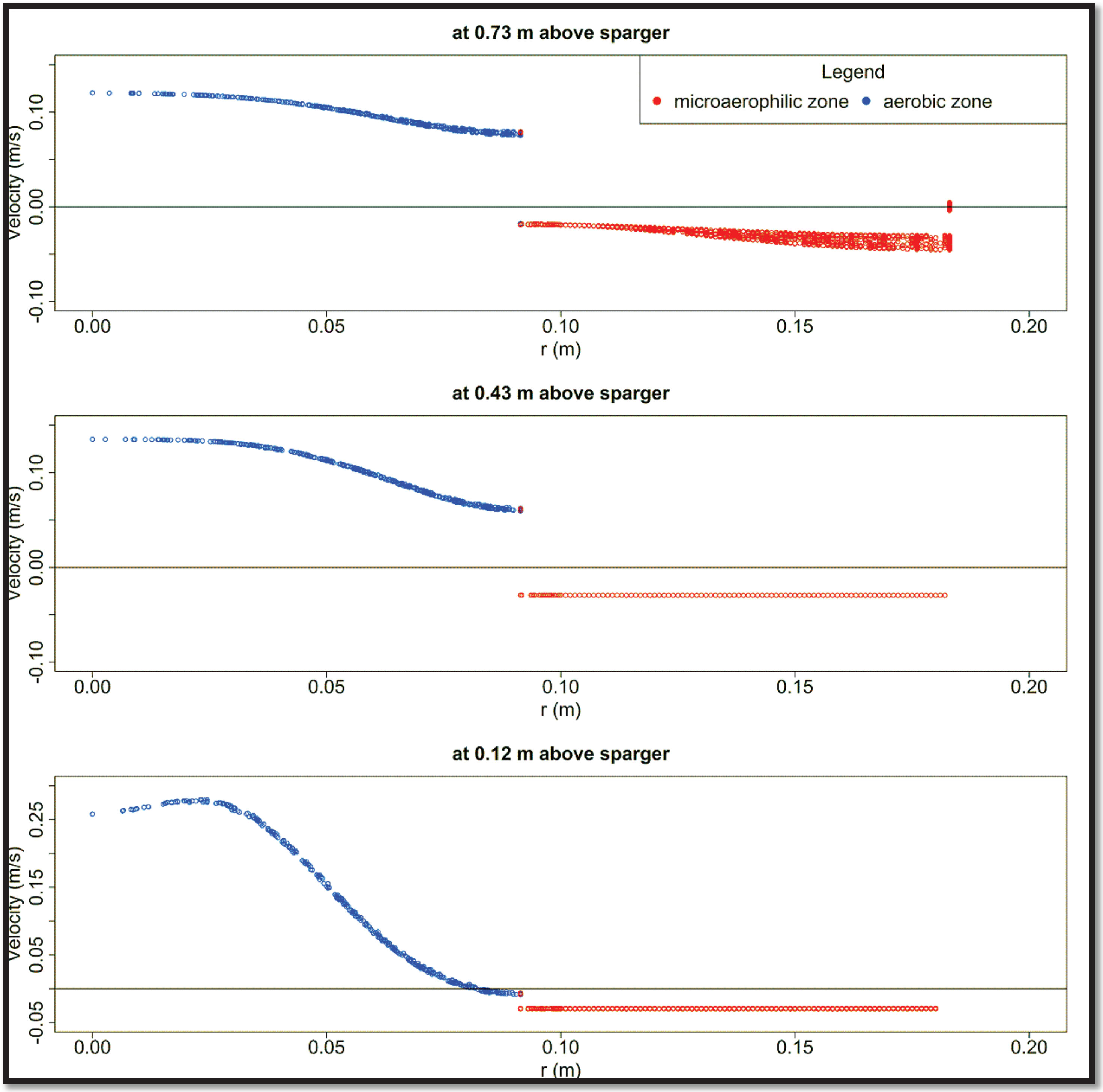


Figure 1.4: Vertical (Z-direction) velocity at three elevations in the bioreactor modelled in Run 22

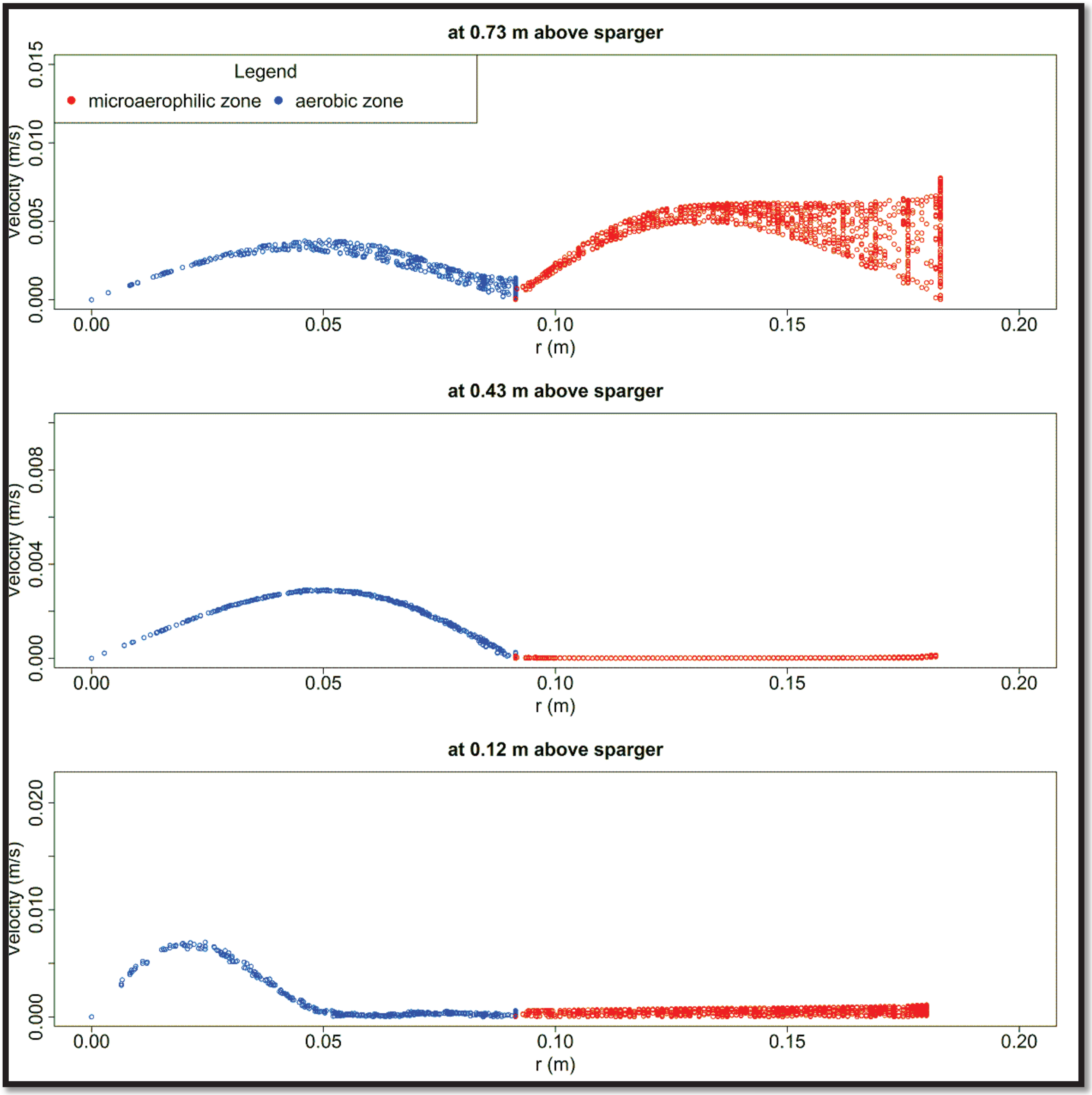


Figure 1.5: X-Y projection of velocity vector in the bioreactor modelled in Run 10

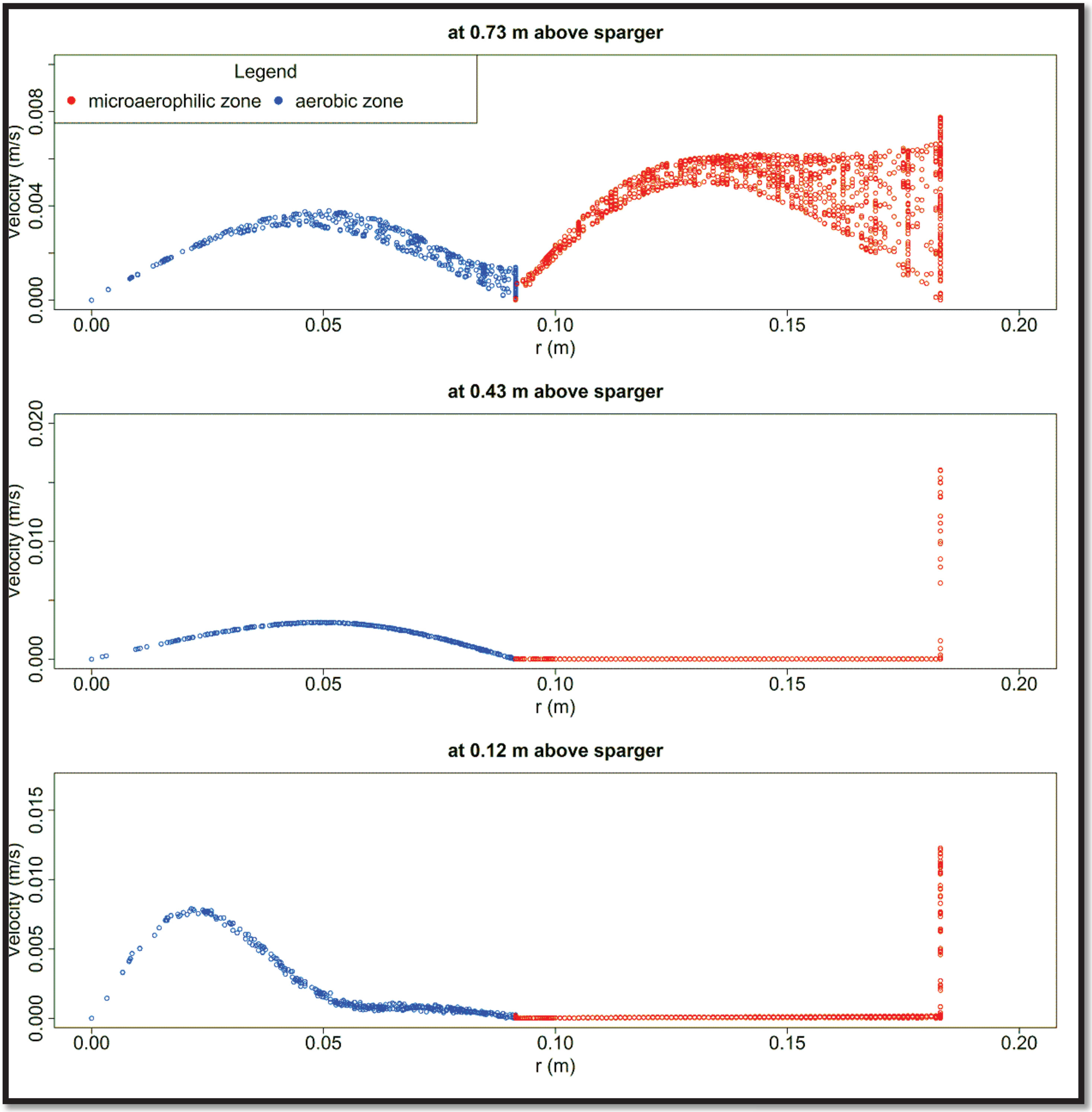


Figure 1.6: X-Y projection of velocity vector in the bioreactor modelled in Run 22

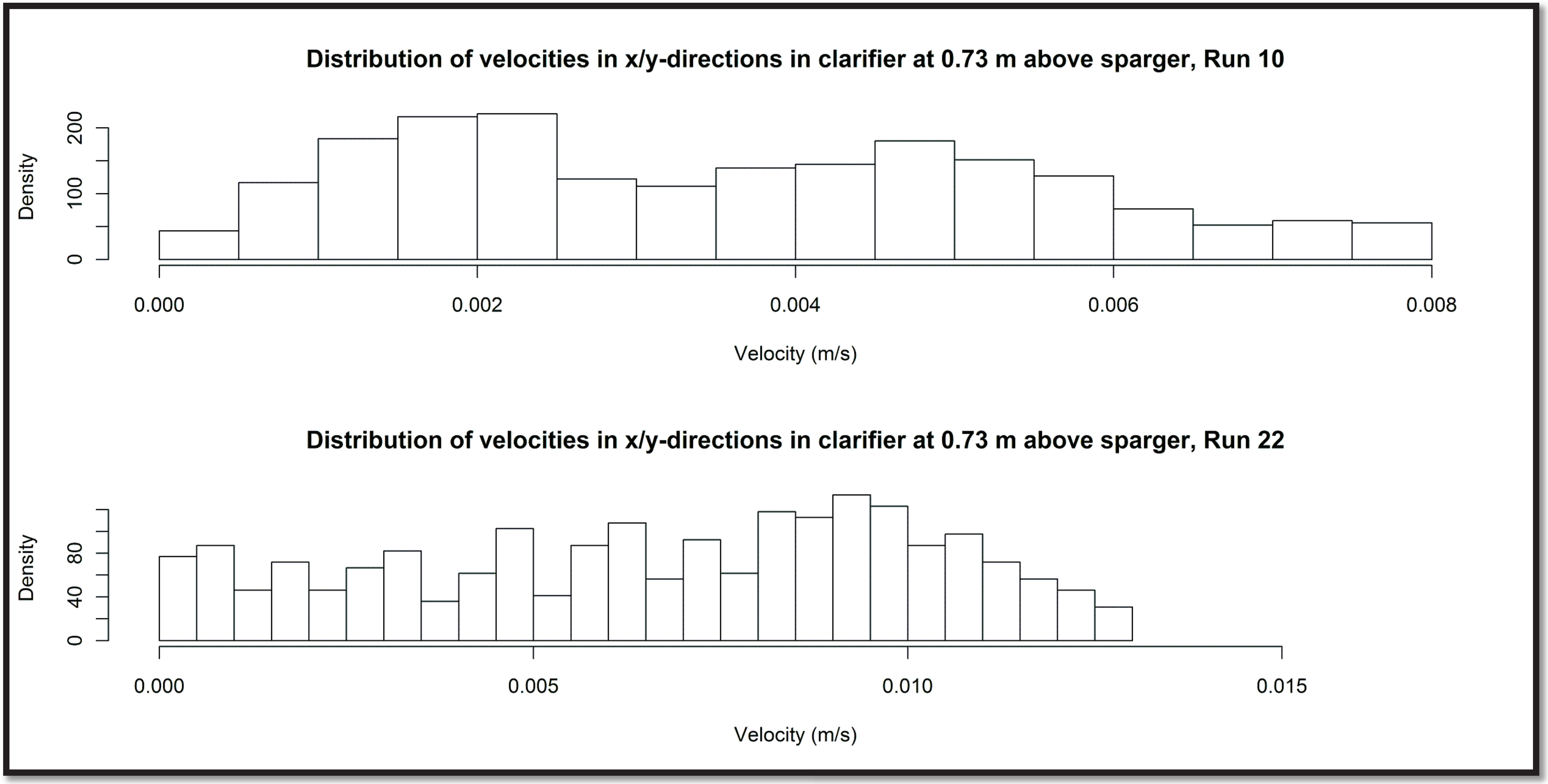
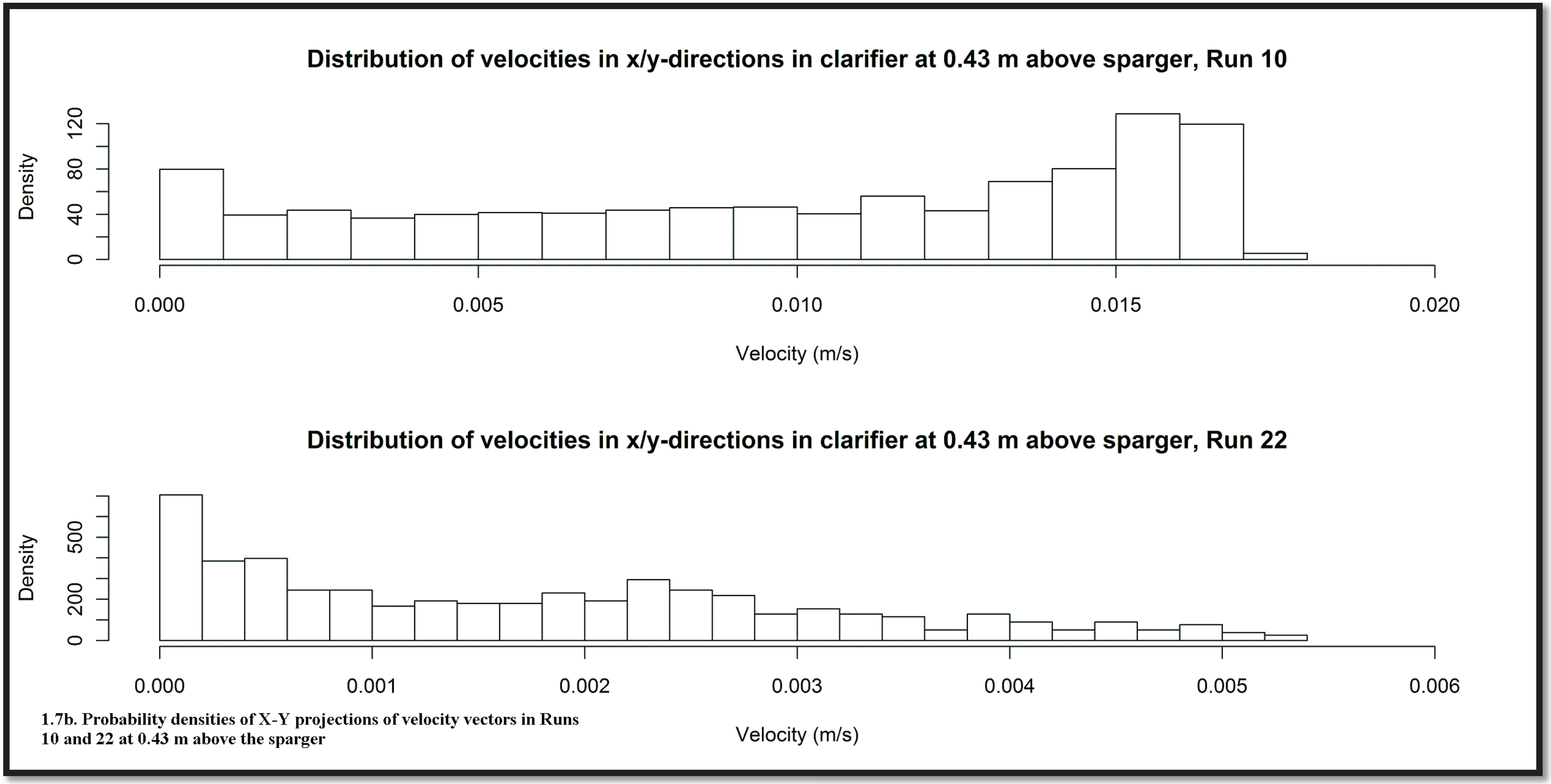


Figure 1.7a: Probability densities of X-Y projections of velocity vectors in Runs 10 and 22 at 0.73 m above the sparger



1.7b. Probability densities of X-Y projections of velocity vectors in Runs 10 and 22 at 0.43 m above the sparger

Figure 1.7b: Probability densities of X-Y projections of velocity vectors in Runs 10 and 22 at 0.43 m above the sparger

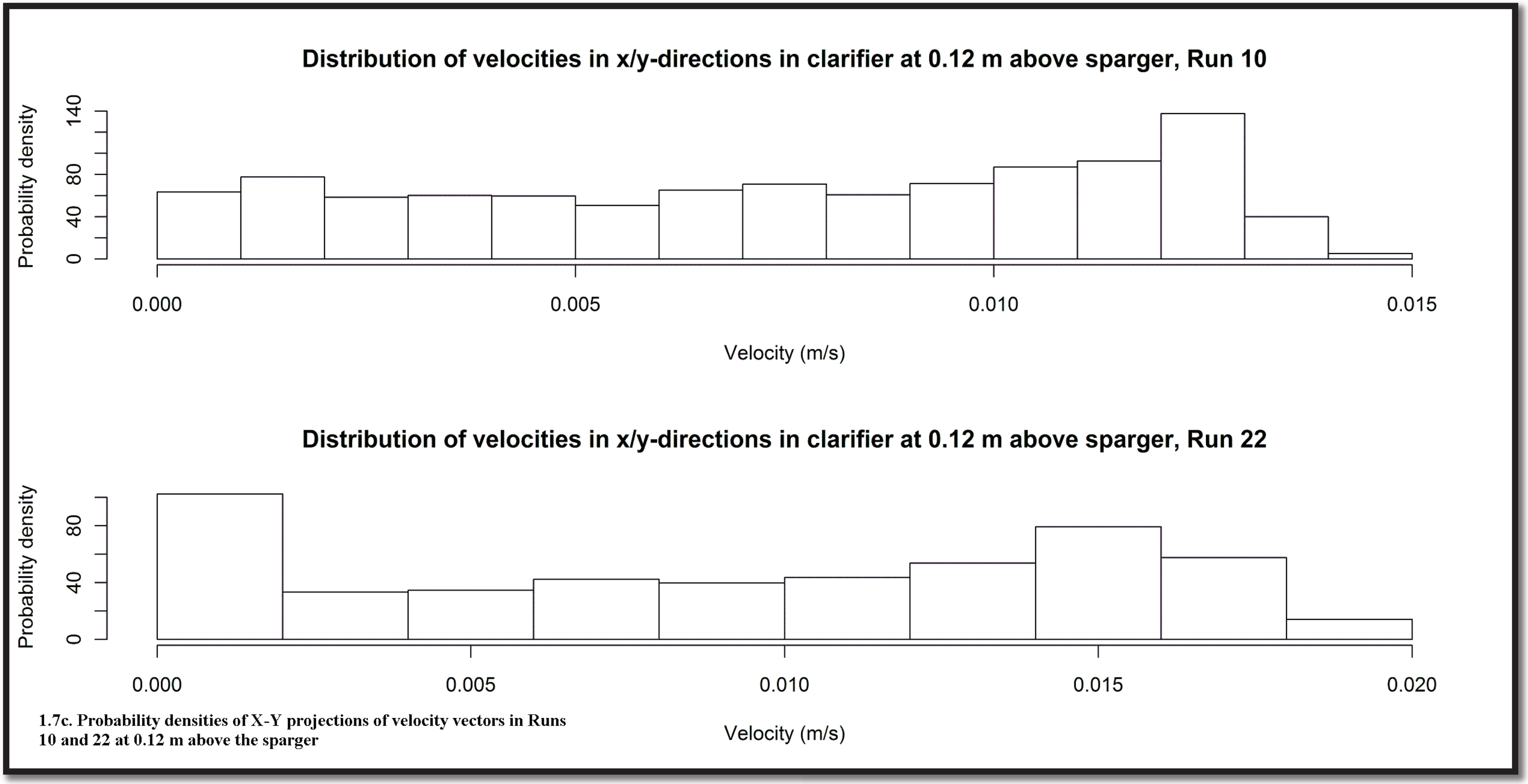


Figure 1.7c: Probability densities of X-Y projections of velocity vectors in Runs 10 and 22 at 0.12 m above the sparger

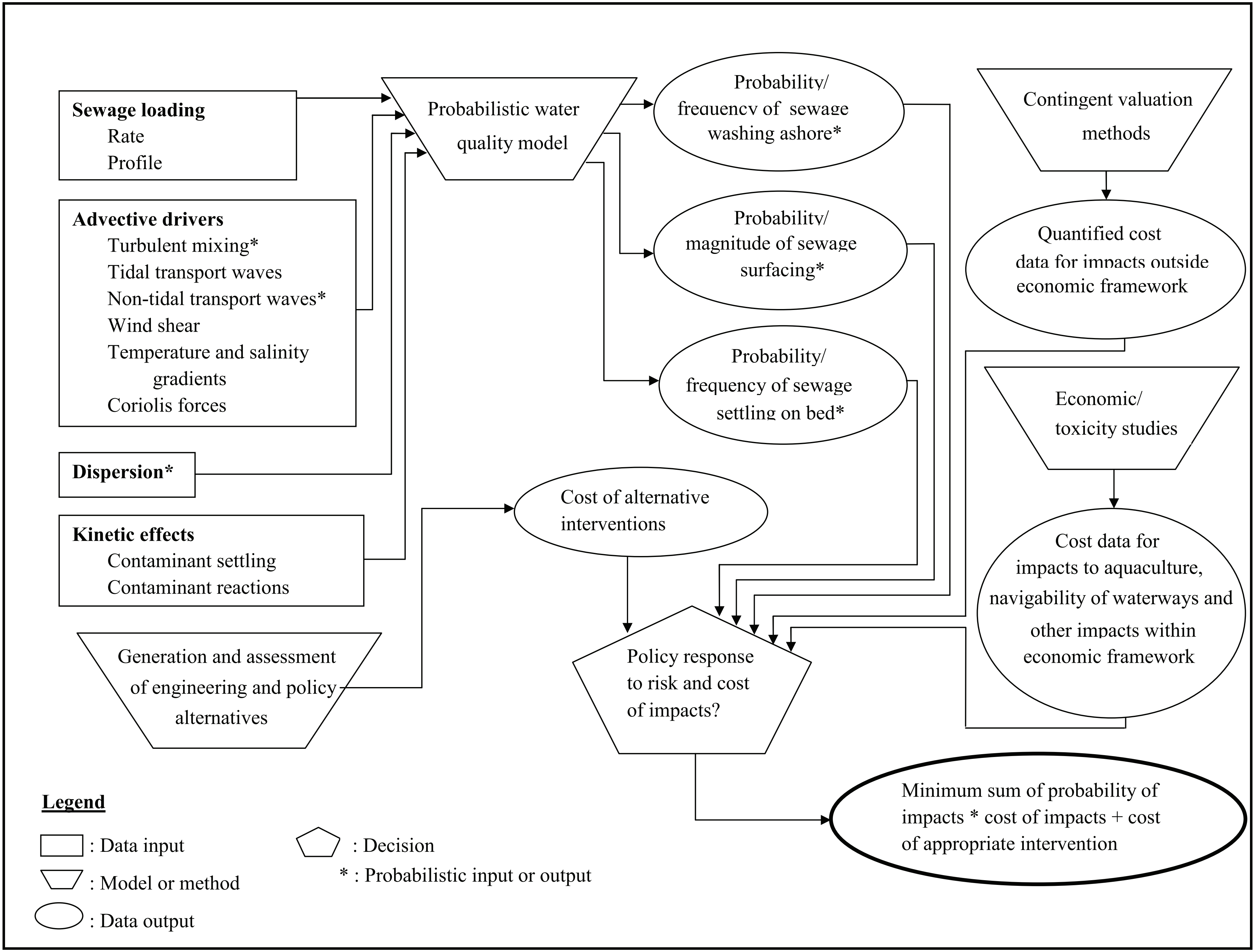


Figure 2.1: Conceptual outline of decision model

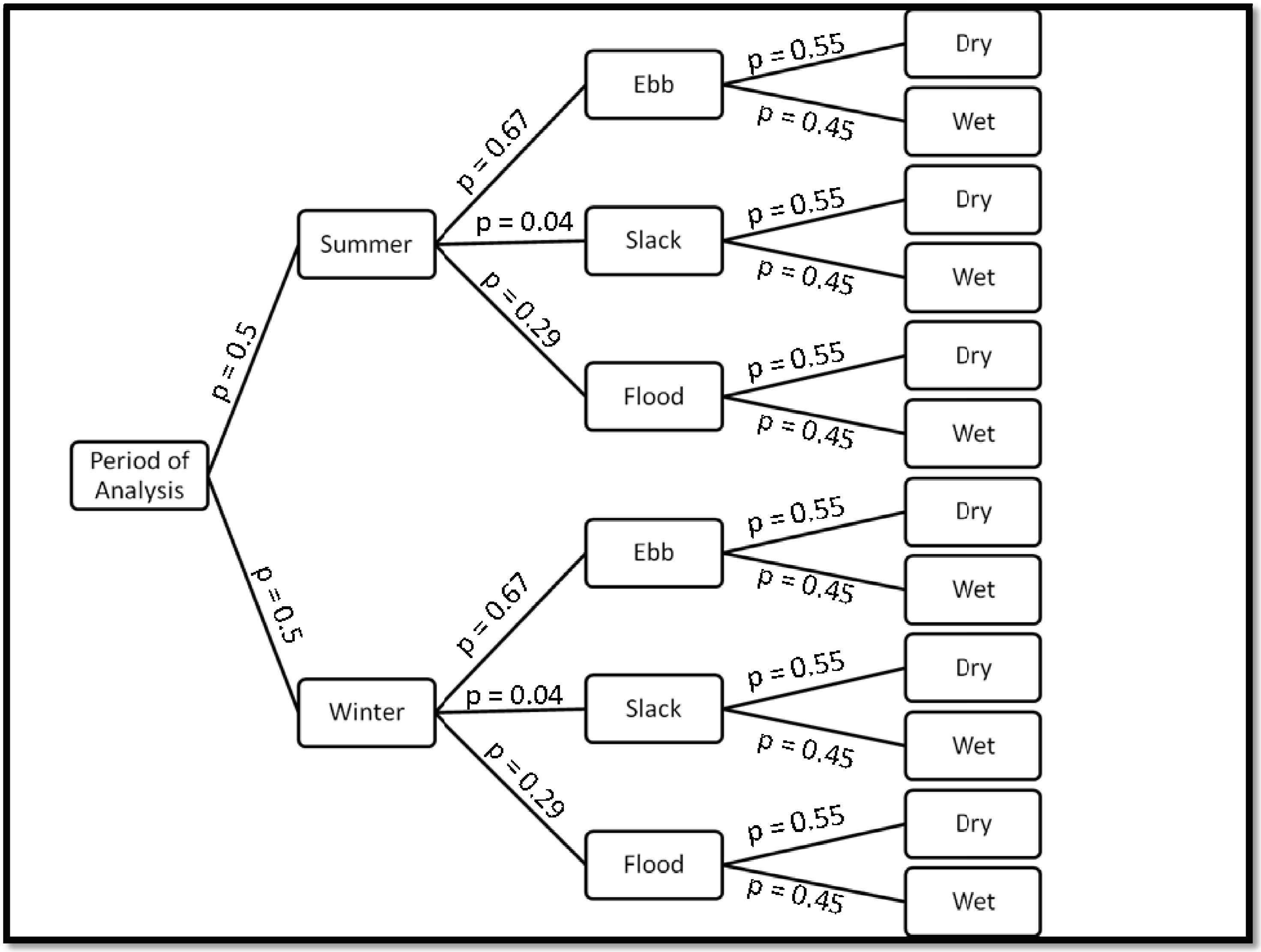


Figure 2.2: Event tree for CFD simulations

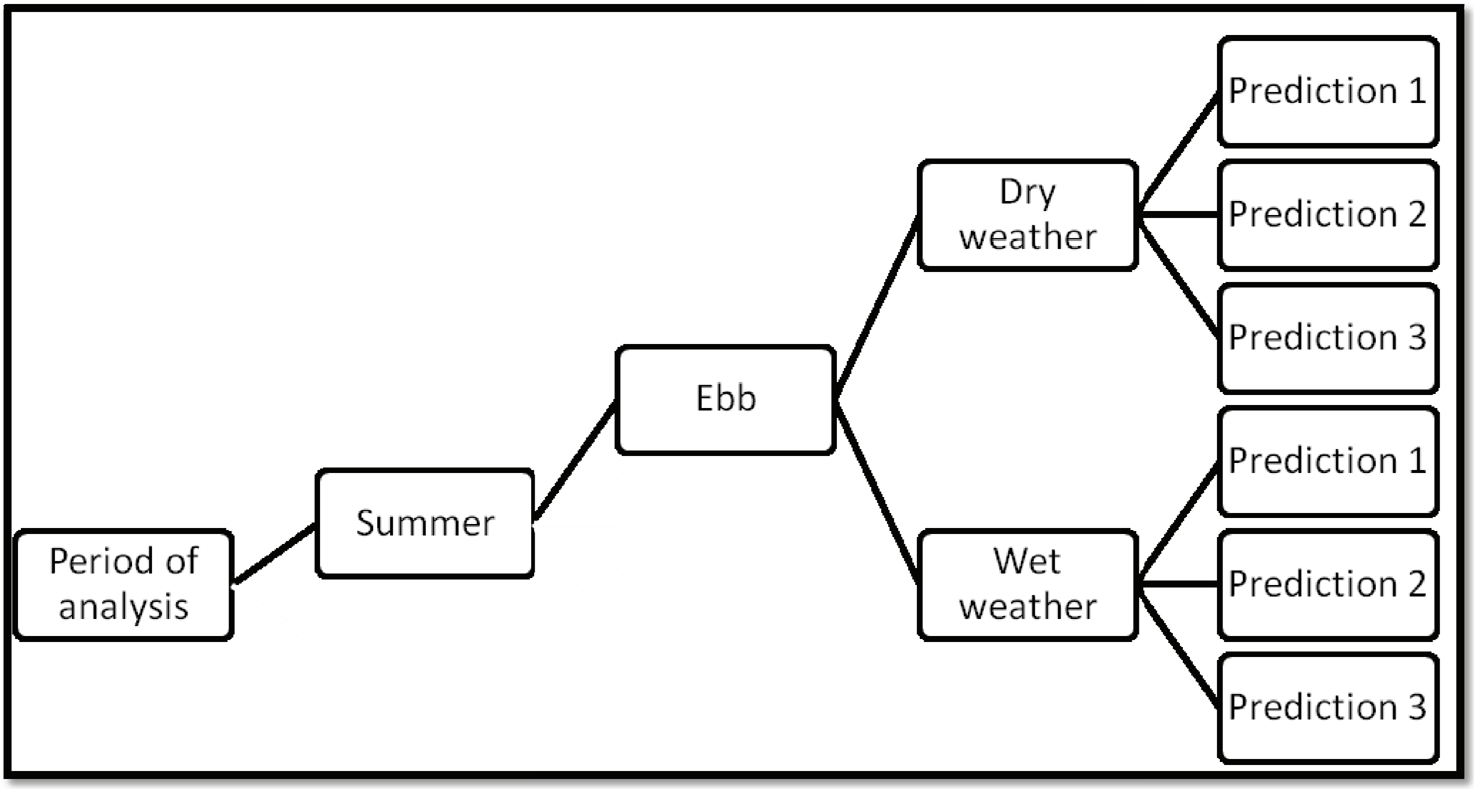


Figure 2.3: Expansion of event tree for multiple predictions of each outcome

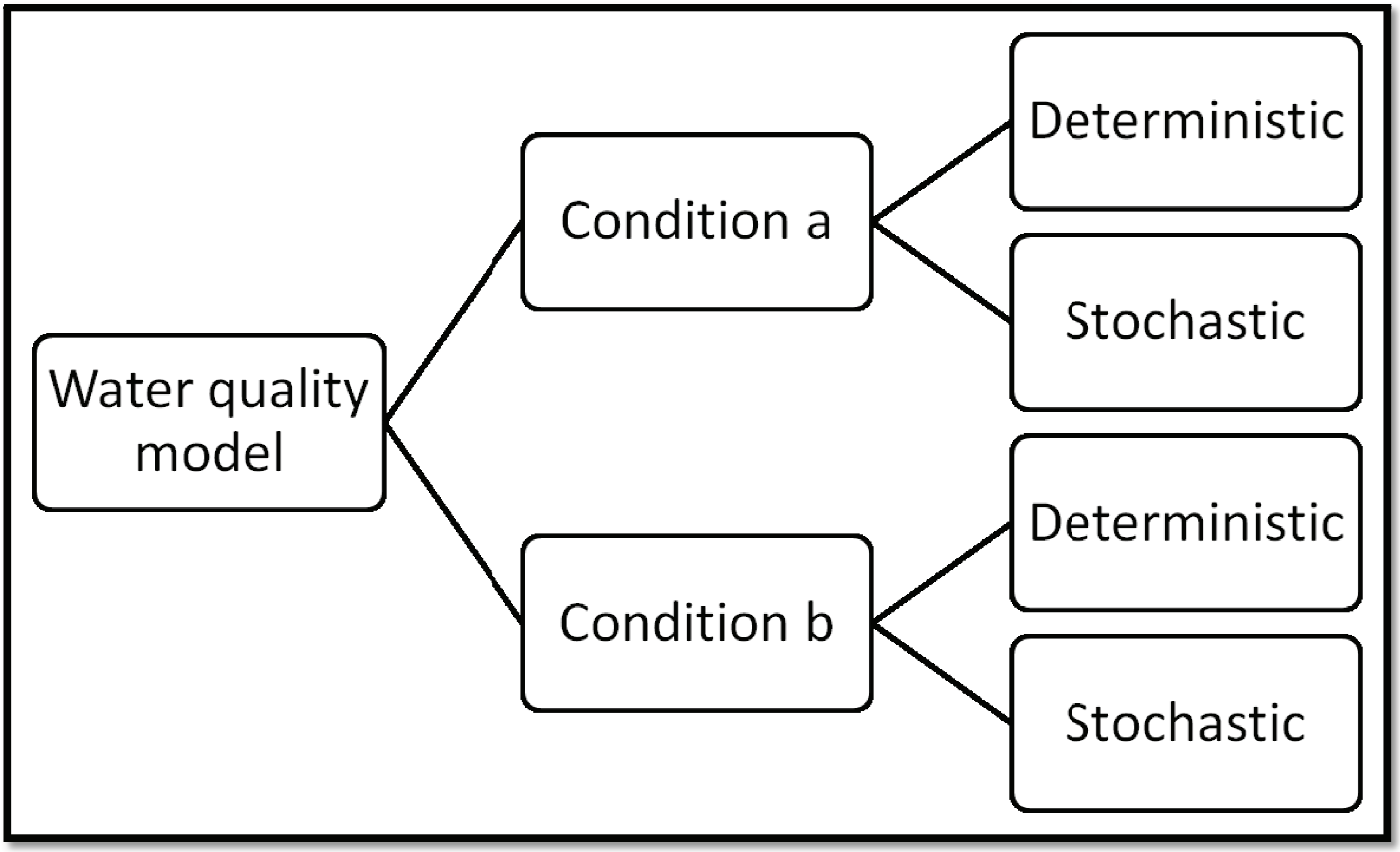


Figure 2.4: Expansion of event tree for probabilistic parameters

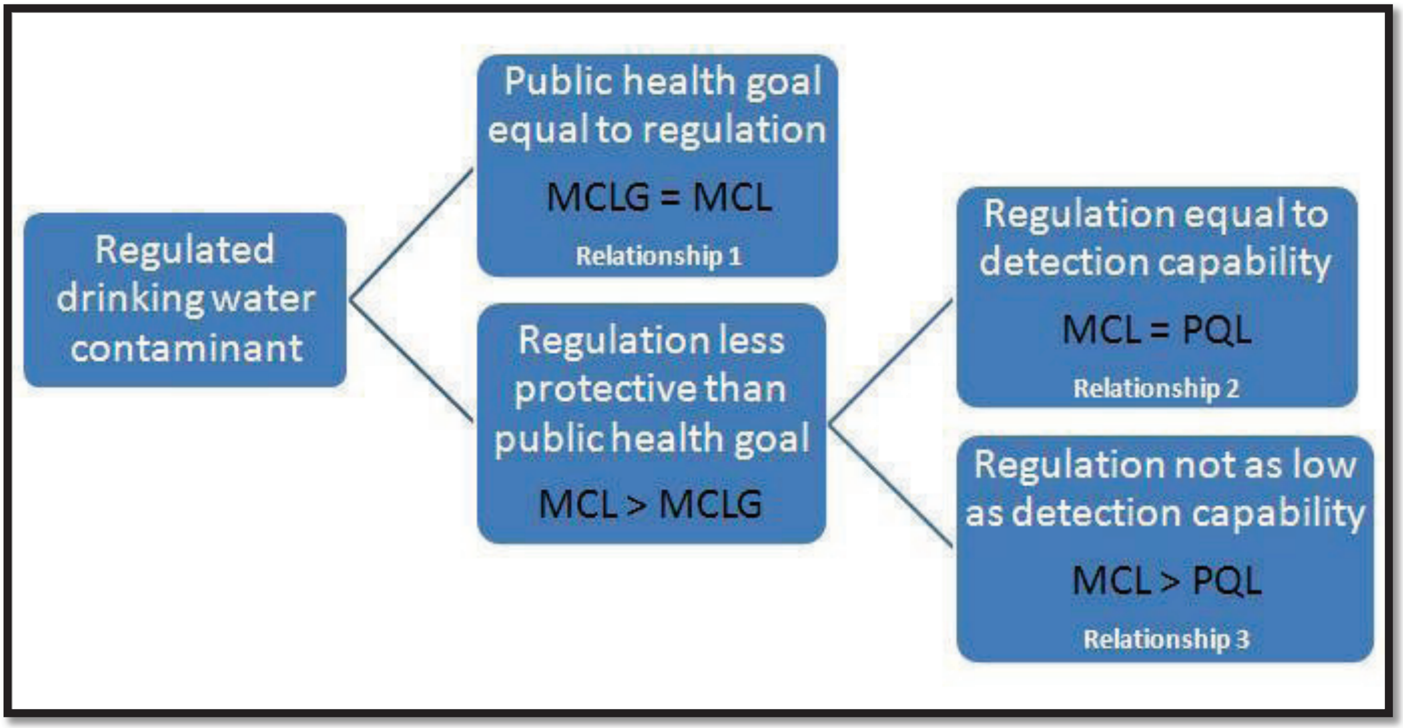


Figure 3.1: Relationships between public health goal (MCLG), regulation (MCL), and detection capability (PQL) in NPDWR contaminants

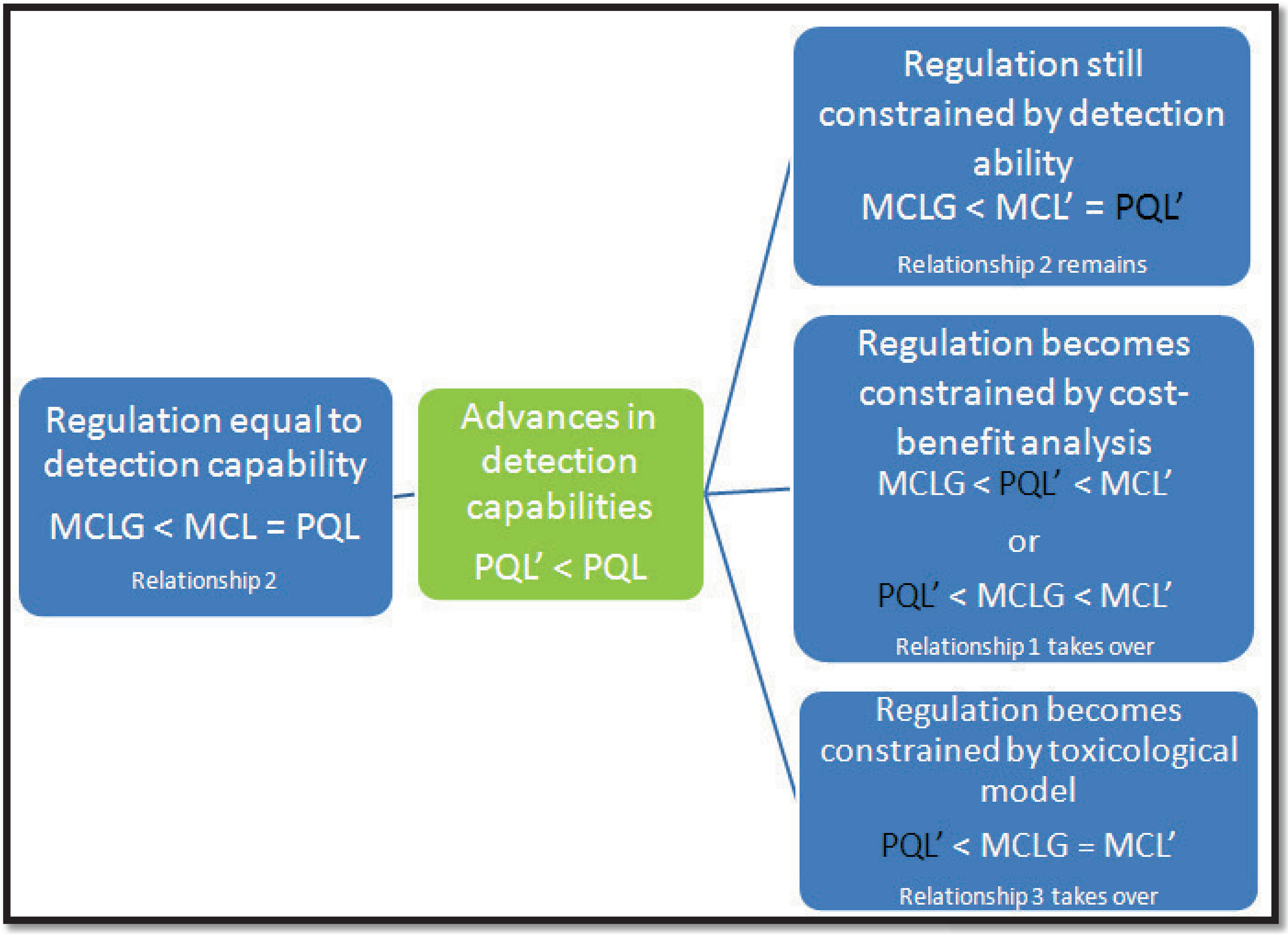


Figure 3.2: The three possible outcomes of a regulation revision triggered by advances in detection abilities

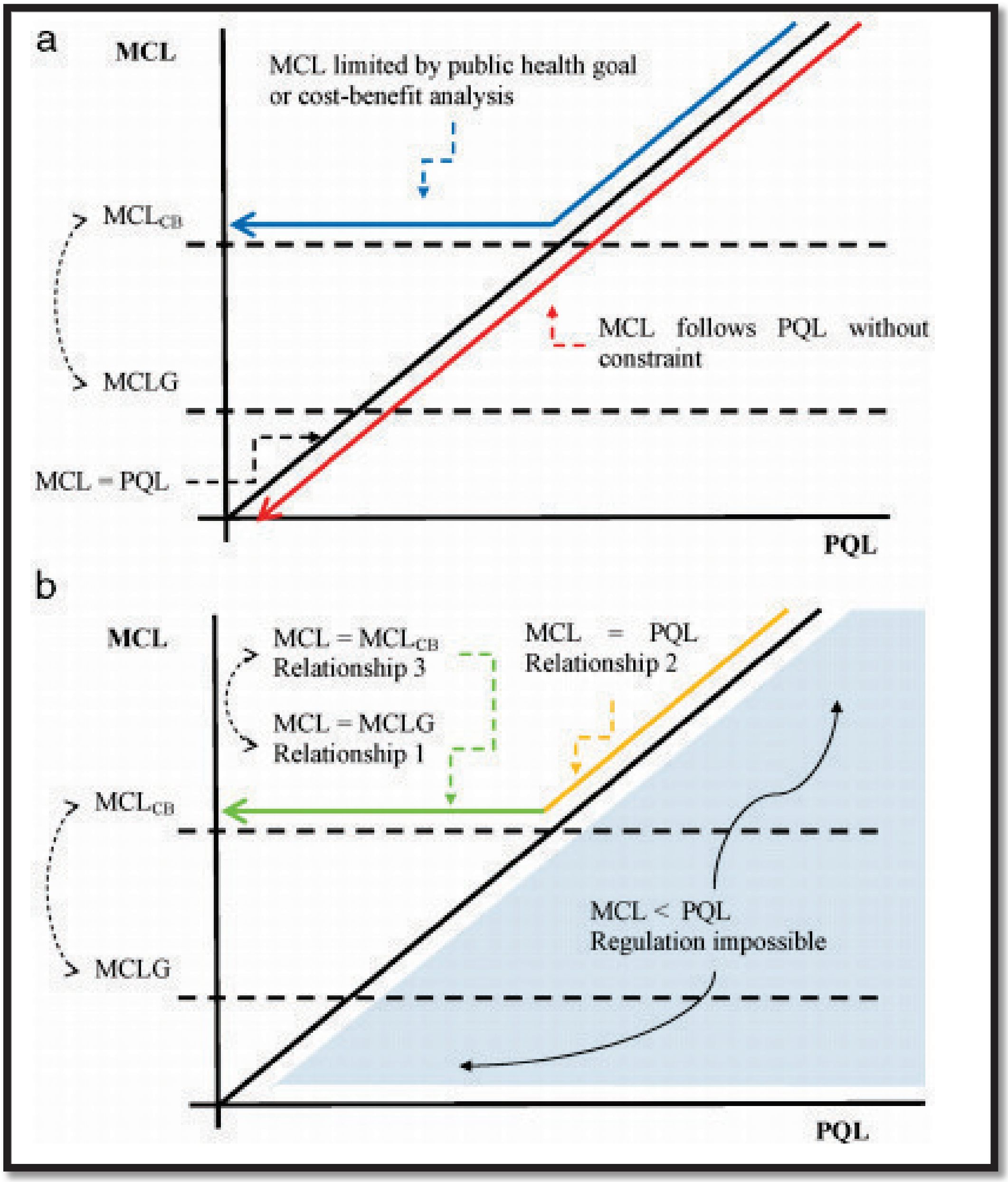


Figure 3.3: Two competing conceptions of how regulations follow detection ability

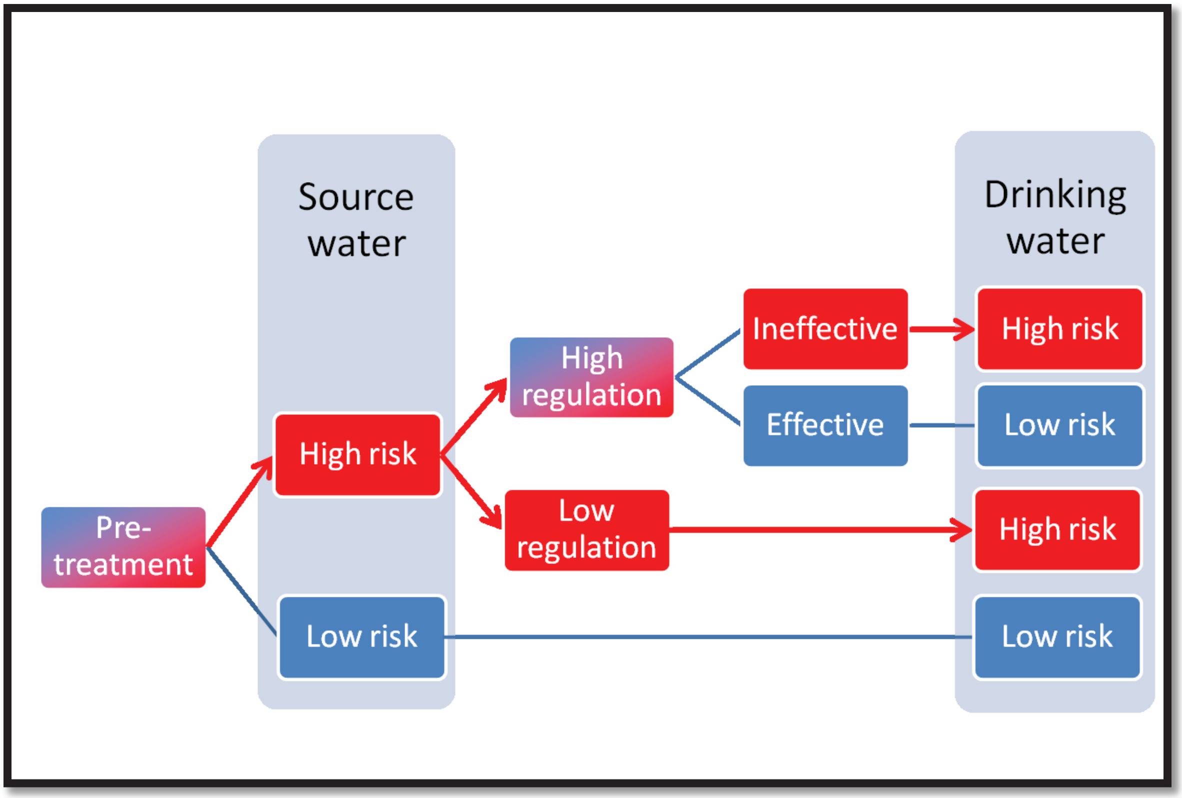


Figure 4.1: Event tree connecting source water quality to drinking water quality through degree and effectiveness of regulation (abridged)

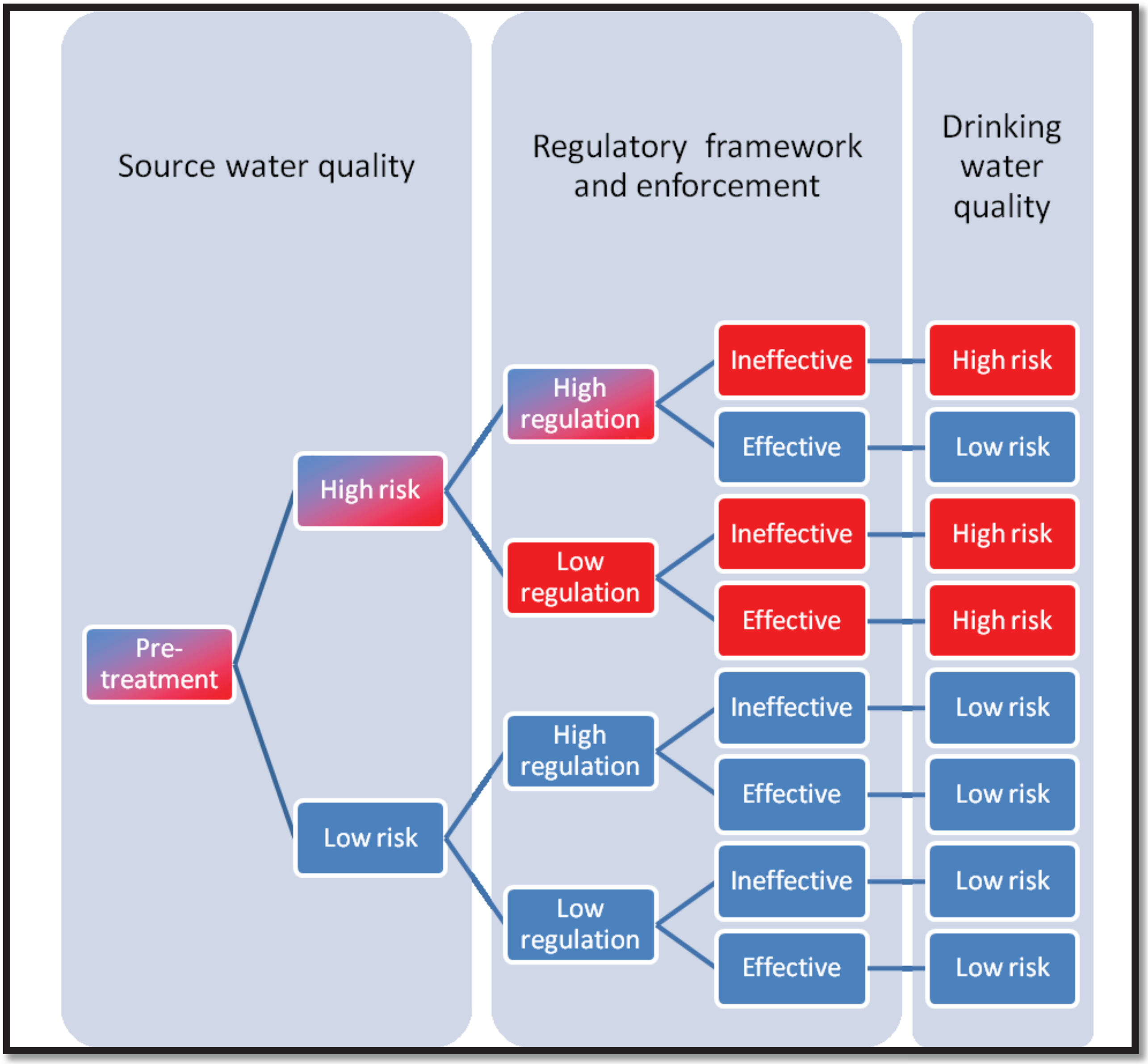


Figure 4.2: Event tree connecting source water quality to drinking water quality through degree and effectiveness of regulation (expanded)

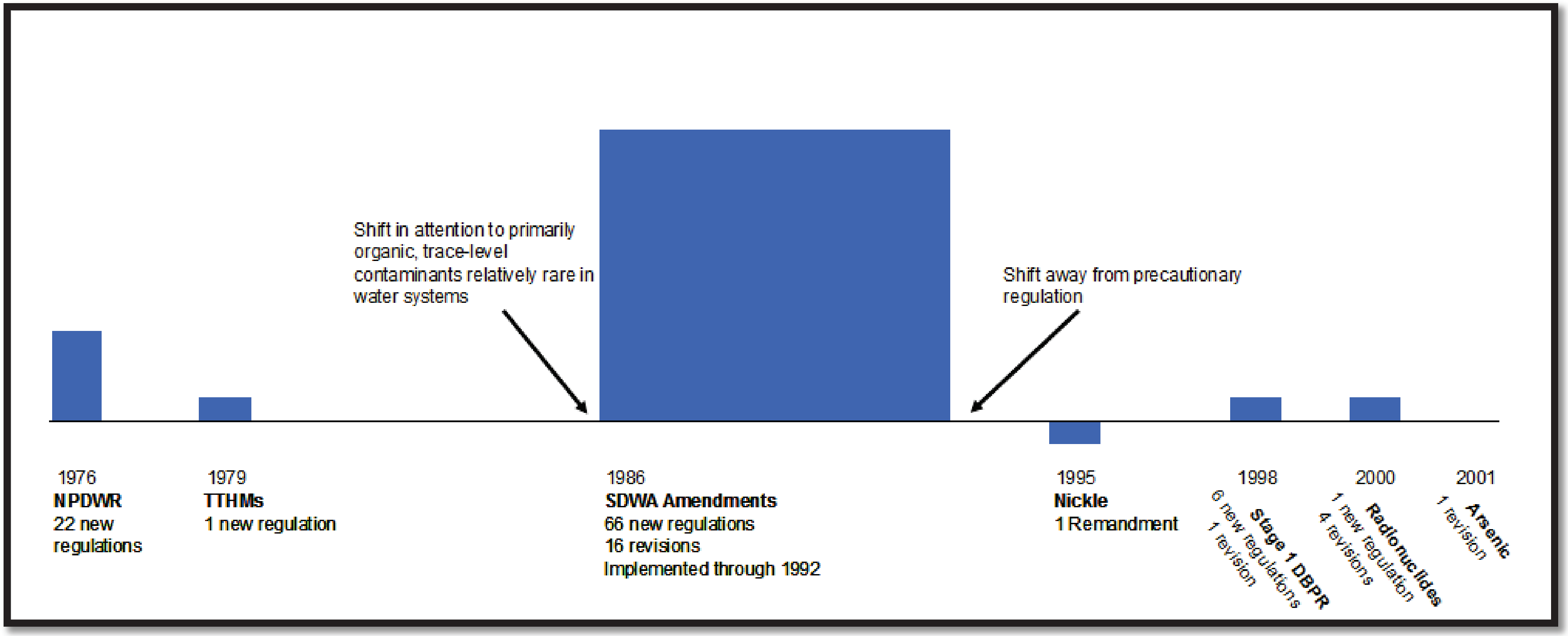


Figure 5.1: Drinking water contaminant regulation timeline

TABLES

Table 1.1: Summary of runs

Run	Skirt		Cone height (cm)	D_{ma} : D_a	Air flow rate (L min^{-1})	Wastewater flow (L d^{-1})	Sparger		Comments
	Length (cm)	Angle (degrees)					Shape	Height of air input* (cm)	
1	8.5	45	18	2	30	70	Cylinder	6	As-built lab scale at "realistic" wastewater flow rate
2	8.5	45	18	2	30	70	Cylinder	2	To test the effect of lowering the air sparger (as in the torus runs)
3	8.5	45	18	2	10	70	Cylinder	6	Runs 3 to 5 test the effects of varying the air inflow rate in the as-built dimensions with respect to Run 1
4	8.5	45	18	2	50	70	Cylinder	6	
5	8.5	45	18	2	70	70	Cylinder	6	
6	8.5	45	18	2	30	0	Cylinder	6	Runs 6 to 11 vary the wastewater throughput at a fixed air inflow rate, from 0 to the rates used in Farnaz's thesis
7	8.5	45	18	2	30	1	Cylinder	6	
8	8.5	45	18	2	30	720	Cylinder	6	
9	8.5	45	18	2	10	70	Torus	2	Runs 9 to 12 are repeats of Runs 1 and 3 to 5, but with a torus-shaped sparger. Run 2 demonstrates that the higher air sparger height of the cylindrical sparger does not produce significant differences in the hydrodynamic characteristics.
10	8.5	45	18	2	30	70	Torus	2	
11	8.5	45	18	2	50	70	Torus	2	
12	8.5	45	18	2	70	70	Torus	2	
13	8.5	45	18	2	30	0	Torus	2	Runs 13 to 18 vary wastewater throughput at a fixed air flow rate in order to find the minimum throughput to eliminate circulation in the clarifier for the torus sparger configuration.
14	8.5	45	18	2	30	1	Torus	2	
15	8.5	45	18	2	30	150	Torus	2	
16	8.5	45	18	2	30	200	Torus	2	
17	8.5	45	18	2	30	500	Torus	2	
18	8.5	45	18	2	30	720	Torus	2	
19	0	NA	18	2	30	70	Torus	2	Runs 19 to 23 vary the geometry of the skirt between zones 2 and 3 under constant conditions
20	15	45	18	2	30	70	Torus	2	
21	4	90	18	2	30	70	Torus	2	
22	12	0	18	2	30	70	Torus	2	
23	8.5	0	18	2	30	70	Torus	2	
24	8.5	45	30	2	30	70	Torus	2	Run 24 varies the height of the bottom cone
25	8.5	45	18	1.5	30	70	Torus	2	Run 25 varies the ratio Dma/Da

*Height of air input surface above wall separating aerobic and microaerophillic zones

D_{ma}: Diameter of microaerophilic zone

D_a: Diameter of aerobic zone

NA: Not applicable

Table 1.2: Turbulence scales

Q (L min^{-1})	$u_{g,s}$ (m s^{-1})	d_b (mm)	ε_g[1]	ε_g[2]	$u_{g,r}$ (m s^{-1})	$u_{g,r\ max}$ (m s^{-1})	τ (s)	T_D (s)	Stokes number	C_{TD}
10	0.008	6.2	0.021	0.024	0.36	3.87	2.52E-03	4.31E-02	5.83E-02	8.87E+00
30	0.023	6.8	0.058	0.059	0.40	3.88	3.00E-03	4.31E-02	6.96E-02	7.36E+00
50	0.038	7.0	0.089	0.091	0.43	3.88	3.25E-03	4.30E-02	7.56E-02	6.74E+00
70	0.053	7.2	0.117	0.115	0.46	3.88	3.43E-03	4.30E-02	7.99E-02	6.35E+00

[1]: Calculated

[2]: Experimental [1]

Table 1.3: CFD model output validation with respect to experiments and reference values

Run	From computational model				Reference values		
	Mean z velocity (m/s)		Circulation time through zones 1 and 2 (s)		Mean z velocity - zone 1 (m/s) Chisti et al, 1988	Mean z velocity - zone 1 (m/s) Bello et al, 1984	Circulation time - experimental (s) Behzadian, 2010
	Zone 1	Zone 2					
1	0.0852	-0.0175	10.73	52.39	0.15	0.50	33
2	NC	NC	NC	NC	0.15	0.50	33
3	0.0501	-0.0108	18.24	84.62	0.10	0.35	67
4	0.1080	-0.0212	8.47	43.15	0.20	0.55	28.5
5	0.1275	-0.0237	7.17	38.59	0.25	0.60	26
6	0.1572	-0.0358	5.82	25.51	0.15	0.50	33
7	0.0857	-0.0177	10.67	51.79	0.15	0.50	33
8	0.0570	-0.0123	16.03	74.16	0.15	0.50	33
9	0.0567	-0.0164	16.13	55.77	0.10	0.35	67
10	0.0826	-0.0229	11.07	39.86	0.15	0.50	33
11	0.1068	-0.0284	8.56	32.18	0.20	0.55	28.5
12	0.1238	-0.0317	7.39	28.87	0.25	0.60	26
13	0.1560	-0.0475	5.86	19.24	0.15	0.50	33
14	0.0843	-0.0235	10.84	38.95	0.15	0.50	33
15	0.1511	-0.0460	6.05	19.87	0.15	0.50	33
16	0.0817	-0.0227	11.19	40.26	0.15	0.50	33
17	0.0854	-0.0238	10.70	38.36	0.15	0.50	33
18	0.0858	-0.0239	10.65	38.27	0.15	0.50	33
19	0.0864	-0.0253	10.58	36.18	0.15	0.50	NA
20	0.0829	-0.0236	11.03	38.68	0.15	0.50	NA
21	NC	NC	NC	NC	0.15	0.50	NA
22	0.0820	-0.0232	11.16	39.48	0.15	0.50	NA
23	0.1328	-0.0408	6.89	23.92	0.15	0.50	NA
24	0.0807	-0.0235	11.34	38.89	0.15	0.50	NA
25	0.0490	-0.0149	18.64	61.52	0.15	0.50	NA

Note: Zones 1 and 2 are the aerobic and microaerophilic zones respectively

*Height of air input surface above wall separating aerobic and microaerophillic zones

D_{ma}: Diameter of microaerophilic zone

D_a: Diameter of aerobic zone

NA: Not applicable

NC: Not calculated

Table 2.1: Risk and sample impact valuation table

From Li & Hodgins 2010								
CFD Run	**Season**	**Tide**	**Weather**	**Marginal probability**	**Plume characteristics**	**Fishing**	**Recreation**	**Property values**
1	Summer	Ebb	Dry	0.18	Acceptable			
2	Summer	Ebb	Wet	0.15	Far transport	x		
4	Summer	Flood	Dry	0.08	Far transport	x		
5	Summer	Flood	Wet	0.07	Far transport, surfacing, settling	x	x	x
7	Summer	Slack	Dry	0.01	Far transport, surfacing, settling	x	x	x
8	Summer	Slack	Wet	0.01	Far transport	x		
10	Winter	Ebb	Dry	0.18	Far transport, surfacing	x	x	x
11	Winter	Ebb	Wet	0.15	Far transport, surfacing	x	x	x
13	Winter	Flood	Dry	0.08	Far transport, surfacing, settling	x	x	x
14	Winter	Flood	Wet	0.07	Far transport, surfacing, settling	x	x	x
16	Winter	Slack	Dry	0.01	Far transport	x		
17	Winter	Slack	Wet	0.01	Far transport, surfacing, settling	x	x	x
					Fractional losses with respect to pollution-free scenario	0.82	0.57	0.06

Table 2
Summary of EPA data for contaminants targeted by the Analytical Feasibility Support Document

Table 3.1: Contaminants sorted by improvement in method sensitivity, PQL re-evaluation and relationship between MCL and MCLG

SDWA Chemical Contaminant	MCLG (mg/L)	MCL (mg/L)	Current PQL (mg/L)	MCL = PQL?	PQL > MCLG?	MCL > MCLG?	Improvement in sensitivity of methods since promulgation	Lowest original MDL (µg/L)	Lowest current MDL (µg/L)	Analytical method with lowest MDL	MDL of most common method (µg/L)	Most common analytical method	Most widely used MDL = lowest MDL available?	Improvement in sensitivity of methods 2000-2007	PQL reevaluation? Review 1	PQL reevaluation? Review 2
	Original contaminants targeted by first Six-Year Review (2003)															
Alachlor	zero	0.002	0.002	yes	yes	yes	no	0.225	0.005	551.5	0.14	507	no	no	(1)	(2)
Benzene	zero	0.005	0.005	yes	yes	yes	no	0.01	0.01	502.2	0.03 - 0.04	524.4	no	no	(2)	(2)
Benzo(a)pyrene	zero	0.0002	0.0002	yes	yes	yes	yes	0.016	0.016	550.1	0.032 - 0.23	525.2	no	no	(1)	(1)
Beryllium	0.004	0.004	0.001	no	no	no	yes	0.02	0.02	200.9	0.02 and 0.3	200.9 and 200.7	half	no	(1)	(1)
Bis(2-ethylhexyl)phthalate	zero	0.006	0.006	yes	yes	yes	yes	0.6 - 0.8*	0.46 - 1.3	525.1	0.46 - 1.3	525.2	yes	no	(1)	(1)
Cadmium	0.005	0.005	0.002	no	no	no	no	0.1	0.03 - 0.5 and 0.05	200.8 and 200.9	1 and 0.05	200.7 and 200.9	half	no	(2)	(1)
Carbofuran	0.04	0.04	0.007	no	no	no	yes	1.5	0.52	531.1	0.52	531.1	yes	yes	(2)	(2)
Carbon tetrachloride	zero	0.005	0.005	yes	yes	yes	no	0.01-0.02*	0.002 - 0.006	551.1	0.08 - 0.21	524.2	no	no	(2)	(2)
Chlordane	zero	0.002	0.002	yes	yes	yes	yes	0.0015	0.001 - 0.004	508.1	0.0016 - 0.0041	508	no	no	(2)	(2)
Chromium (Cr III and VI)	0.1	0.1	0.1	yes	no	no	no	0.6	0.08 - 0.9	200.8	4	200.7	no	no	(1)	(1)
1,2-Dibromo-3-chloropropane (DBCP)	zero	0.0002	0.0002	yes	yes	yes	no	0.01	0.006 - 0.009	551.1	0.01	504.1	no	no	(2)	(2)
1,4-Dichlorobenzene (para)	0.075	0.075	0.005	no	no	no	no	0.006	0.01 - 0.04	502.2	0.03 - 0.04	524.2	no	no	(2)	(2)
1,2-Dichloroethane	zero	0.005	0.005	yes	yes	yes	no	0.002	0.02 - 0.06	524.2	0.02 - 0.06	524.2	yes	no	(2)	(2)
1,1-Dichloroethylene	0.007	0.007	0.005	no	no	no	no	0.003	0.04 - 0.10	502.2	0.05 - 0.12	524.2	no	no	(2)	(2)
Dichloromethane (methylene chloride)	zero	0.005	0.005	yes	yes	yes	no	0.01 - 0.02	0.01 - 0.02	502.2	0.03 - 0.09	524.2	no	no	(2)	(1)
1,2-Dichloropropane	zero	0.005	0.005	yes	yes	yes	no	0.01 - 0.03	0.01 - 0.03	502.2	0.02 - 0.04	524.2	no	no	(2)	(2)
Dioxin (2,3,7,8-TCDD)	zero	3	0.00000003	no	yes	yes	no	1.00E-08	1.00E-08	1613	1.00E-08	1613	yes	no	(1)	N/A
Diquat	0.02	0.02	0.004	no	no	no	yes	0.44	0.72	549.2	0.72	549.2	yes	no	(1)	(1)
Ethylene dibromide	zero	0.00005	0.00005	yes	yes	yes	no	0.01	0.007 - 0.008	551.1	0.01	504.1	no	no	(1)	(1)
Fluoride	4	4	0.5	no	no	no	no	0.05 - 1.5	0.01	300	N/A	Manual electrode	yes	no	(1)	(1)
Glyphosate	0.7	0.7	0.06	no	no	no	no	5.99 - 8.99	5.99 - 8.99	547	5.99 - 8.99	547	yes	no	(1)	(1)
Heptachlor	zero	0.0004	0.0004	yes	yes	yes	yes	0.003	0.0015	508	0.0015	508	yes	no	(2)	(2)
Heptachlor epoxide	zero	0.0002	0.0002	yes	yes	yes	no	0.004	0.001	508.1	0.0059	508	no	no	(2)	(2)
Hexachlorobenzene	zero	0.001	0.001	yes	yes	yes	no	0.002	0.001 - 0.003 and 0.002	551.1 and 505	0.0077	508	no	no	(2)	(2)
Hexachlorocyclopentadiene	0.05	0.05	0.001	no	no	no	no	0.03 - 0.1	0.072 - 0.16	525.2	0.072 - 0.16	525.2	yes	no	(2)	(2)
Mercury	0.002	0.002	0.0005	no	no	no	no	0.2	0.2	245.1	0.2	245.1	yes	no	(1)	(1)
Methoxychlor	0.04	0.04	0.01	no	no	no	no	0.04 - 0.3	0.008 - 0.023	551.1	0.022	508	no	no	(2)	(2)
Oxamyl (Vydate)	0.2	0.2	0.02	no	no	no	no	2	0.86	531.1	0.86	531.1	yes	no	(2)	(1)
PCBs (as decachlorobiphenyl)	zero	0.0005	0.0005	yes	yes	yes	no	0.08 - 0.23	0.08 - 0.23	508A	0.08 - 0.23	508A	yes	no	(1)	(1)
Pentachlorophenol	zero	0.001	0.001	yes	yes	yes	no	0.076	0.021 - 0.085 and 0.032	515.3 and 515.1	0.032	515.1	yes	yes	(1)	(1)
Picloram	0.5	0.5	0.001	no	no	no	no	0.14	0.15	515.1	0.15	515.1	yes	no	(1)	(1)
Tetrachloroethylene	zero	0.005	0.005	yes	yes	yes	no	0.001	0.002	551.1	0.05 - 0.14	524.2	no	no	(2)	(2)
Thallium	0.0005	0.002	0.002	yes	yes	yes	no	0.3	0.01 - 0.3	200.8	0.7	200.9	no	no	(2)	(1)
Toluene	1	1	0.005	no	no	no	no	0.01 - 0.02	0.01 - 0.02	502.2	0.08 - 0.11	524.2	no	no	(2)	(2)
Toxaphene	zero	0.003	0.003	yes	yes	yes	yes	1	0.13	508.1	ND	508	no	no	(2)	(2)
1,1,1-Trichloroethane	0.2	0.2	0.005	no	no	no	yes	0.003	0.005 - 0.012	551.1	0.04 - 0.08	524.2	no	yes	(2)	(2)
1,1,2-Trichloroethane	0.0003	0.005	0.005	yes	yes	yes	yes	0.007	0.012 - 0.014	551.1	0.03 - 0.1	524.2	no	no	(2)	(2)
Trichloroethylene	zero	0.005	0.005	yes	yes	yes	no	0.001	0.002 - 0.004	551.1	0.02 - 0.19	524.2	no	no	(2)	(2)
Vinyl chloride	zero	0.002	0.002	yes	yes	yes	no	0.01	0.01 - 0.18	502.2	0.01 - 0.18	502.2	yes	no	(1)	(2)
Xylene (total)	10	10	0.005	no	no	no	no	0.002 - 0.004	0.01 - 0.02	502.2	0.03 - 0.13	524.2	no	no	(2)	(1)
	Additional contaminants targeted by second Six-Year Review (2009)															
Antimony	0.006	0.006	0.006	yes	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
Arsenic	zero	0.01	0.003	no	yes	yes	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Atrazine	0.003	0.003	0.001	no	no	no	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(2)
Barium	2	2	0.15	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
Bromate	zero	0.01	0.01	yes	yes	yes	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(1)
Copper	1.3	TT	0.05	no	no	N/A	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Chlorobenzene	0.1	0.1		no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
Cyanide	0.2	0.2	0.1	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
2,4-D	0.07	0.07	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(1)
Dalapon	0.2	0.2	0.01	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Dichlorobenzene (ortho)	0.6	0.6	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
cis-1,2-Dichloroethylene	0.07	0.07	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
trans-1,2-Dichloroethylene	0.1	0.1	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
Di(2-ethylhexyl) adipate	0.4	0.4	0.006	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Dinoseb	0.02	0.02	0.002	no	no	no	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(1)
Endothall	0.1	0.1	0.09	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)

Table 2
Summary of EPA data for contaminants targeted by the Analytical Feasibility Support Document

SDWA Chemical Contaminant	MCLG (mg/L)	MCL (mg/L)	Current PQL (mg/L)	MCL = PQL?	PQL > MCLG?	MCL > MCLG?	Improvement in sensitivity of methods since promulgation	Lowest original MDL (µg/L)	Lowest current MDL (µg/L)	Analytical method with lowest MDL	MDL of most common method (µg/L)	Most common analytical method	Most widely used MDL = lowest MDL available?	Improvement in sensitivity of methods 2000-2007	PQL reevaluation? Review 1	PQL reevaluation? Review 2
Endrin	0.002	0.002	0.0001	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Ethylbenzene	0.7	0.7	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
Lead	zero	TT	0.005	no	yes	N/A	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(1)
Lindane	0.0002	0.0002	0.0002	yes	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
Nitrate	10	10	0.4	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Nitrite	1	1	0.4	no	no	no	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(2)
Selenium	0.05	0.05	0.01	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Simazine	0.004	0.004	0.01	no	yes	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(1)
Styrene	0.1	0.1	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)
2,4,5-TP (Silvex)	0.05	0.05	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	yes	N/A	(2)
1,2,4-Trichlorobenzene	0.07	0.07	0.005	no	no	no	ND	ND	ND	ND	ND	ND	ND	no	N/A	(2)

(1) Current PQL still appropriate or insufficient data to reach conclusion
(2) Conclusions of Six-Year Review support change or may support change of PQL
ND: Not determined
MCLG < MCL = PQL

TABLE 3.2: Tabulation of contaminants sorted by improvement in method sensitivity, PQL re-evaluation and relationship between MCL and MCLG for first and second Six-Year Reviews (2003 and 2009), as presented in table 3.1

Contaminants targeted by both Six-Year Reviews (2003 and 2009) as presented in Review 1 (2003)*

Improvement in sensitivity of methods since promulgation	PQL reevaluation indicated by Six-Year Review?	Count MCLG < MCL = PQL (Relationship 2)	% of total count	Count PQL < MCL (Relationships 1 or 3)	% of total count
yes	yes	3	14	2	11
yes	no	2	9	2	11
no	yes	12	55	8	44
no	no	5	23	6	33

Contaminants targeted by both Six-Year Reviews (2003 and 2009) as presented in Review 2 (2009)

Improvement in sensitivity of methods 2000-2007	PQL reevaluation indicated by Six-Year Review?	Count MCLG < MCL = PQL (Relationship 2)	% of total count	Count PQL < MCL (Relationships 1 or 3)	% of total count
yes	yes	3	14	2	12
yes	no	3	14	2	12
no	yes	12	55	5	29
no	no	4	18	8	47

Additional contaminants targeted by second Six-Year Review (2009) as discussed in Review 2 (2009)

Improvement in sensitivity of methods 2000-2007	PQL reevaluation indicated by Six-Year Review?	Count MCLG < MCL = PQL (Relationship 2)	% of total count	Count PQL < MCL** (Relationships 1 or 3)	% of total count
yes	yes	0	0	3	12
yes	no	1	50	3	12
no	yes	1	50	7	28
no	no	0	0	12	48

* Includes dioxin, which was not targeted by Review 2 (2009)

** Includes copper and lead, two contaminants that have prescribed treatment efficacies instead of MCLs but that are not limited by detection ability

Table 3.3: Historical regulatory revisions of chemical contaminants

SDWA Chemical Contaminant	Initial standard (year)	MCL (year)	MCLG (current)	PQL (current)	Notes
2,4,5-TP	0.01 mg/L (1975); Proposed MCL = MCLG = 0.05 mg/L (1989)	0.05 mg/L (1992)	0.05 mg/L	0.05 mg/L	Value of standard **increased** between 1975 and 1989. 1989 value adopted in 1992.Not identified as possibly subject to a toxicological review.
2,4-D	0.1 mg/L (1975)	0.07 mg/L (1991)	0.07 mg/L	0.05 mg/L	Not identified as possibly subject to a toxicological review.
Arsenic	NIPDWR: 0.05 mg/L	0.01 mg/L (2001)	zero	0.03 mg/L	Regulation lowered from 0.05 to 0.01 mg/L on account of growing body of evidence suggesting old standard was inadequate to protect health. Note that reg > PQL. Question of cost/benefit.
Barium	USPHS: 1.0 mg/L (1962) NIPDWR: 1.0 mg/L; Proposed MCL = MCLG = 5 mg/L (1989);	2 mg/L (1991)	2 mg/L	0.15 mg/L	Value of standard has **increased** since 1962. Not identified as possibly subject to a toxicological review.
Cadmium	0.01 mg/L (1975)	0.005 mg/L (1991)	0.005 mg/L	0.002 mg/L	Standard lowered, but this followed a revision to the concentration considered to be protective of public health, not a significant improvement in detection ability
Chromium	0.05 mg/L (1975)	0.1 mg/L (1991)	0.1 mg/L	0.1 mg/L	Value of standard has **increased**.
Endrin	0.0002 mg/L (1975)	0.002 mg/L	0.002 mg/L	0.0001 mg/L	Value of standard has **increased**.
Fluoride	1.4 to 2.4 mg/L (1975)	4 (1986)	4	0.5 mg/L	Standards have been decided upon based on various health studies with variable importance being ascribed to sensitive populations and "cosmetic" impacts. Value of standard has
Lead	0.05 mg/L (1975)	(Treatment technology)	zero	0.005 mg/L	Lead regulation has changed from a maximum concentration to a mandatory removal efficiency
Lindane	0.004 mg/L (1975)	0.0002 mg/L	0.0002 mg/L	0.0002 mg/L	Value of standard has **decreased**. No information found tying re-regulation to detection capabilities
Methoxychlor	0.1 mg/L (1975)	0.04 mg/L (1992)	0.04 mg/L	0.01 mg/L	Regulation has been lowered, but not to PQL. Therefore, toxicological model or cost-benefit analysis dominates the regulation.
Nickel	0.1 mg/L (1992)	N/A (1995)	0.1	N/A	Regulation **remanded**
Selenium	0.01 mg/L (1975)	0.05 mg/L (1991)	0.05 mg/L	0.001 mg/L	Value of standard has **increased**.
THMs	0.1 mg/L (1979)	0.8 (1998)	N/A	N/A	Value of standard has **increased**.THMs are regulated only to the extent to which implementation would not result in exceedences to microbiological drinking water standards. Regulations are function of competing risk models.
Toxaphene	0.005 mg/L (1975)	0.003 mg/L (1991)	zero	0.003 mg/L (1991)	Toxaphene MCL is tied to PQL.

Table 5.1: Water quality findings of CWSS

Population ranges for water system groups											
Population range in thousands		**< 0.5**	**0.5 to 1**	**1 to 5**	**5 to 10**	**10 to 25**	**25 to 50**	**50 to 100**	**100 to 1000**	**> 1000**	**Total**
Total population served in thousands		87.60	73.00	532.30	539.60	1018.60	1380.20	1110.00	5362.50	8100.00	18203.8
Number of systems		446.00	101.00	214.00	75.00	59.00	36.00	16.00	21.00	1.00	969
Water systems (percent)											
Population range in thousands		**< 0.5**	**0.5 to 1**	**1 to 5**	**5 to 10**	**10 to 25**	**25 to 50**	**50 to 100**	**100 to 1000**	**> 1000**	
Met 1962 standards		50.0	67.0		64.0	71.0	67.0	62.0	73.0		N/A
Violated standards (total)		50.0	33.0		36.0	29.0	33.0	38.0	27.0		N/A
Violated standards (recommended)		26.0	21.0		25.0	20.0	30.0	31.0	27.0		N/A
Violated standards (mandatory)		24.0	12.0		11.0	9.0	3.0	7.0	0.0		N/A
Total population (thousands)											
Met 1962 standards	**Average**	43.8	405.6		345.3	723.2	924.7	688.2	9827.6		12958.5
	Low	0.0	105.5		240.0	418.9	603.0	496.0	1606.0		3469.4
Violated standards (total)	**Average**	43.8	199.7		194.3	295.4	455.5	421.8	3634.9		5245.3
	Low	0.0	52.0		135.0	171.1	297.0	304.0	594.0		1553.1
Violated standards (recommended)	**Average**	22.8	127.1		134.9	203.7	414.1	344.1	3634.9		4881.5
	Low	0.0	33.1		93.8	118.0	270.0	248.0	594.0		1356.8
Violated standards (mandatory)	**Average**	21.0	72.6		59.4	91.7	41.4	77.7	0.0		363.8
	Low	0	18.9		41.25	53.1	27	56	0		196.25
Total population (percent)											
Met 1962 standards	**Average**	50.0	67.0		64.0	71.0	67.0	62.0	73.0		71.2
	Low	0.0	17.4		44.5	41.1	43.7	44.7	11.9		19.1
Violated standards (total)	**Average**	50.0	33.0		36.0	29.0	33.0	38.0	27.0		28.8
	Low	0.0	8.6		25.0	16.8	21.5	27.4	4.4		8.5
Violated standards (recommended)	**Average**	26.0	21.0		25.0	20.0	30.0	31.0	27.0		26.8
	Low	0.0	5.5		17.4	11.6	19.6	22.3	4.4		7.5
Violated standards (mandatory)	**Average**	24.0	12.0		11.0	9.0	3.0	7.0	0.0		2.0
	Low	0.0	3.1		7.6	5.2	2.0	5.0	0.0		1.1

Table 5.2: Contaminants covered by 1962 PHS and 1974 NIPDWR regulations and CWSS findings

Contaminant	1962 PHS reccomended	% systems exceeding	% population exceeding	1962 PHS mandatory	% systems exceeding	% population exceeding	NPDWR	Comments
2,4-D	NA	NA	NA	NA	NA	NA	0.1	Changed in 1991 to 0.07 mg/L
2,4,5-TP (Silvex)	NA	NA	NA	NA	NA	NA	0.01	Changed in 1991 to 0.05 mg/L
Alkyl benzene sulfonate	0.5	0	NR	NA	NA	NA	NA	
Arsenic	0.01	< 1	NR	0.05	< 1	NR	0.05	Changed in 2001 to 0.01 mg/L
Barium	NA	NA	NA	1	< 1	NR	1	Changed in 1991 to 2 mg/L
Boron[1]	NA	1	NR	NA	0	0	NA	
Cadmium	NA	NA	NA	0.01	< 1	NR	0.01	Changed in 1991 to 0.005 mg/L
Chloride	250	1	NR	NA	NA	NA	NA	
Chromium (VI)	NA	NA	NA	0.05	< 1	NR	0.05	Changed in 1991 to 0.1 mg/L
Coliform bacteria	NA	NA	NA	Note 2	12	NR	Note 2	Changed in 1989 to 0 CFU/100 mL
Copper	1	1	NR	NA	NA	NA	NA	Reinstated in 1991 at 1.3 mg/L
Carbon chloroform extract	0.2	1	1	NA	NA	NA	NA	
Cyanide	0.01	0	NR	0.2	0	0	NA	Reinstated in 1992 at 0.2 mg/L
Endrin	NA	NA	NA	NA	NA	NA	0.0002	Changed in 1992 to 0.002 mg/L
Fluoride	Note 3	5	NR	Note 4	2	NR	Note 5	Changed in 1986 to 4 mg/L
Gross beta	NA	NA	NA	1000 pCi/L	0	0	NA	*2000
Iron	0.3	10	4	NA	NA	NA	NA	
Lead	NA	NA	NA	0.05	1	2	0.05	Changed in 1991 to mandatory removal efficiency
Lindane	NA	NA	NA	NA	NA	NA	0.004	Changed in 1991 to 0.0002 mg/L
Manganese	0.05	9	8	NA	NA	NA	NA	
Mercury	NA	NA	NA	NA	NA	NA	0.002	
Methoxychlor	NA	NA	NA	NA	NA	NA	0.1	*Changed in 1991 to 0.04 mg/L
Nitrate	45	3	1	NA	NA	NA	10	
Phenols	0.001	NA	NA	NA	NA	NA	NA	
Radium 226	3 pCi/L	1	NR	NA	NA	NA	NA	Changed in 2000 to 5 pCi/L including Radium-228
Selenium	NA	NA	NA	0.01	1	NR	0.01	Changed in 1991 to 0.05 mg/L
Silver	NA	NA	NA	0.05	0	0	0.05	Remanded in 1991
Sulfate	250	3	NR	NA	NA	NA	NA	
Strontium-90	10 μμc/L	0	NR	NA	NA	NA	NA	
Total dissolved solids	500	10	3	NA	NA	NA	NA	
Turbidity	5 NTU	3	1	NA	NA	NA	1 to 5	Turbidity requirements updated in 1989 and 1998
Toxaphene	NA	NA	NA	NA	NA	NA	0.005	*Changed in 1991 to 0.003 mg/L
Zinc	5	< 1	NR	NA	NA	NA	NA	

[1] Proposed limit for inclusion in NIPDWR of 1 mg/L

[2] Less than 1 CFU/100 mL for average of all samples AND fewer than 5% of samples > 4 CFU/100 mL

[3] Varies as a function of avg. annual max daily temp. from 0.3 to 1.7 mg/L

[4] Varies as a function of avg. annual max daily temp. from 1.4 to 2.4 mg/L

[5] Varies as a function of avg. annual max daily temp. from 0.4 to 2.4 mg/L

NA: Not applicable
NR: Not reported

Table 5.3: NOMS and NORS findings in finished drinking water (1977)					
Contaminant	**Water systems with detections**[1]	**Median (μg/L)**[2]	**95th percentile (μg/L)**[2]	**Maximum (μg/L)**	**Regulatory level (μg/L)**
Total trihalomethanes (NOMS)	113	61.0	234.00	550.0	100[3]
Total trihalomethanes (NORS)	79	42.1	204.1	482.0	
Dichloroiodomethane	87	NR	NC	NR	NA
1,2-dichloroethane (NOMS)	4	1.53	1.97	2.00	5
1,2-dichloroethane (NORS)	4	ND[4]	0.40	6.0	
Carbon tetrachloride (NOMS)	17	1.26	10.78	23.0	5
Carbon tetrachloride (NORS)	4	ND[5]	NC	3.0	
Methylene chloride	15	`	21.40	41.0	NA
Vinyl chloride	2	0.14	0.18	0.18	2
1,1,2-Trichloroethylene	37	0.25	2.42	27.50	5
Tetrachloroethylene	46	0.65	2.20	3.10	5
1,1,1-Trichloroethane	16	1.38	1.72	1.73	200
Bis (2-chloroethyl) ether	20	0.030	0.31	0.360	NA
Bis (2-chloroisopropyl) ether	14	0.10	0.35	0.55	NA
Benzene	9	0.30	1.46	1.50	5
p-Dichlorobenzene	36	0.029	0.91	1.557	75
m-Dichlorobenzene	2	0.11	0.17	0.18	NA
o-Dichlorobenzene	3	0.18	8.21	9.10	600
1,2,4-Trichlorobenzene	9	0.03	6.22	10.00	70
2,4-Dichlorophenol	59	0.05	1.00	10.00	NA
Pentachlorophenol	86	0.05	0.21	0.70	1
PCBs	5	0.10	0.51	0.60	0.5
Fluoroanthene	20	0.013	0.04	0.080	NA
3,4-Benzofluoranthene	0	NA	NA	NA	NA
1,12-Benzoperylene	0	NA	NA	NA	NA
3,4-Benzopyrene	0	NA	NA	NA	NA
Indeno (1,2,3-cd) pyrene	0	NA	NA	NA	NA

[1] NOMS total = 113; NORS total = 80. All data NOMS unless otherwise indicated

[2] Percentile data for NORS is for all data points and for NOMS only for the population of detections

[3] Changed in 1998 to 80 μg/L

[4] 67.5% below detection limit

[5] 87.5% below detection limit

NA: Not applicable
NC: Not calculated
ND: Not detected above detection limit
NR: Not reported

Table 5.4: Occurrence data for regulations 1998 to 2001 and perchlorate

Contaminant	Median[1]	95th percentile[1]	Maximum[1]	Regulatory level[1]	Regulation
Bromate	NC	> 0.03	> 0.03	0.01	Stage 1 DBPR
Chloramine	NR	NR	NR	4	
Chlorine dioxide	NR	NR	NR	4	
Chlorite	0.4 to 0.5	> 1	> 1	1	
Haloacetic acids (5)	0.01 to 0.02	0.06 to 0.07	> 0.100	0.06	
Total THMs	0.03 to 0.04	0.09 to 0.1	> 0.130	0.08	
Cryptosporidium (oocysts/100 L)	0	10 to 99	1 000 to 9 999	TT	Interim ESTWR
Giardia (cysts/100 mL)	0	100 to 999	1 000 to 9 999	TT	
Turbidity (NTU)	NR	NR	NR	TT	
Gross alpha (pCi/L)	NR	NR	NR	15	Radionuclides
Gross beta (mrem/L)	NR	NR	> 5	5	
Radium 226 and 228 (pCi/L)	< 1.65[2]	6.71[2]	> 5	5	
Uranium	< 0.03	< 0.03	> 0.03	0.03	
Arsenic[3]	< 0.001	0.005 to 0.010	0.118	0.01	Arsenic

[1] In mg/L unless otherwise specified in contaminant column

[2] Sum of interpolated values for Radium 226 and 228

[3] Based on occurrence data for community water systems

TT: Treatment technology

NA: Not applicable

NC: Not calculated

ND: Not detected above detection limit

NR: Not reported

Table 5.5: Occurrence data for regulatory analyses for contaminants from CCL1, CCL2 and perchlorate

Contaminant	CCL	HRL (µg/L)	Population exposed to levels above the HRL[1]			
			Round 1 data	Round 2 data	UCMR 1 data	NIRS data
Carcinogenic						
1,3-dichloropropene	2	0.4	2003000	703000	NA	NA
1,1,2,2-tetrachloroethane	2	0.4	4047000	166000	NA	NA
2,4-dinitrotoluene	2	0.05	38000	NA	NA	NA
2,6-dinitrotoluene	2	0.05	0	NA	NA	NA
Aldrin	1	0.002	NA	1052000	NA	NA
DDE	2	0.2	NA	NA	18000	NA
Dieldrin	1	0.002	NA	793000	NA	NA
Hexachlorobutadiene	1	0.9	781000	10000	NA	NA
Noncarcinogenic						
Acanthamoeba	1	NA	NA	NA	NA	NA
Boron	2	1400	NA	NA	NA	372000
DCPA	2	70	NA	NA	113000	NA
EPTC	2	175	0	0	NA	NA
Fonofos	2	10	0	0	NA	NA
Manganese	1	300	NA	NA	NA	2256000
Metribuzin	1	91	NA	0	NA	NA
Napthalene	1	140	11000	0	NA	NA
Sodium	1	120[2]	NA	NA	NA	7147000
Sulfate	1	500000	NA	1918000	NA	NA
Terbacil	2	90	NA	NA	0	NA
Perchlorate		1	NA	NA	> 16 600 000	
		47	NA	NA	110 000 - 400 000	

[1] National extrapolation based on sample occurrence data, as calculated by EPA for all contaminants but perchlorate.
1 and 47 µg/L are the low and high HRLs for different populations developed by EPA (76 FR 7762)
Perchlorate population exposed to levels > 47 µg/L estimated from examination of UCMR 1 data.
Perchlorate population exposed to levels > 1 µg/L estimate from data in 76 FR 7762. Estimating exposure at this level is problematic because it is below the minimum reporting level of 4 µg/L. The estimate we have provided for 1 µg/L is merely that it is greater than the upper estimate for exposure to 4 µg/L.

[2] Concentration that may be "problematic". Not HRL per se.

Table 5.6: Risk data for carcinogenic contaminants

Carcinogenic contaminant evaluated under the CCL	CCL	HRL (µg/L)	Population exposed to levels above the HRL[1]			95th percentile doses (µg/L)			
			UCM Round 1 data	UCM Round 2 data	UCMR 1 data	Round 1 data	Round 2 data	UCMR 1 data	UCMR 1 data
1,3-dichloropropene	2	0.4	2003000	703000	NA	2	30	NA	NA
1,1,2,2-tetrachloroethane	2	0.4	4047000	166000	NA	155	3	NA	NA
2,4-dinitrotoluene[2]	2	0.05	38000	NA	NA	333	NA	NA	NA
2,6-dinitrotoluene	2	0.05	0	NA	NA	0	NA	NA	NA
Aldrin	1	0.002	NA	1052000	NA	NA	3	NA	NA
DDE[2]	2	0.2	NA	NA	17670	NA	NA	17670	17670
Dieldrin	1	0.002	NA	793000	NA	NA	4	NA	NA
Hexachlorobutadiene	1	0.9	781000	10000	NA	8	1	NA	NA

Regulated carcinogenic contaminant	Regulated under	Unit risk (per µg/L)	Regulatory level (µg/L)	Population above the regulatory level at time of regulation (µg/L)	95th percentile dose (µg/L)	Risk at dose equal to regulatory level	Risk at 95th percentile dose
Arsenic	NPDWR	5.00E-05	0.05	5 265 374 to 37 060 079		2.50E-06	
Arsenic	2001 Arsenic rule	5.00E-05	0.01	11 700 000	0.005 to 0.010	5.00E-07	2.50E-07 to 5.00E-07
Acrylamide	1986 Amendments	1.34E-04	TT				
Epichlorohydrin	1986 Amendments	2.80E-07	TT				
Benzene[3]	1986 Amendments	1.60E-06	5	0	1.46E+00	8.00E-06	2.34E-06
		4.40E-07	5	0	1.46E+00	2.20E-06	6.42E-07
Benzo(a)pyrene	1986 Amendments	2.10E-04	0.2			4.20E-05	
Bromate	Stage 1 DBPR	2.00E-05	10		> 30	2.00E-04	6.00E-04
Carbon tetrachloride	1986 Amendments	3.70E-06	5		10.8	1.85E-05	4.00E-05
Chlordane	1986 Amendments	1.00E-05	2			2.00E-05	
1,2-Dichloroethane	1986 Amendments	2.60E-06	5	0	1.97E+00	1.30E-05	5.12E-06
Dichloromethane	1986 Amendments	2.10E-07	5			1.05E-06	
Di(2-ethylhexyl) adipate	1986 Amendments	3.40E-08	400			1.36E-05	
Di(2-ethylhexyl)phthalate	1986 Amendments	4.00E-07	6			2.40E-06	
Dioxin (2,3,7,8-TCDD)	1986 Amendments	3.10E-07	0.00003			9.30E-12	
Ethylene dibromide	1986 Amendments	6.00E-05	0.005			3.00E-07	
Heptachlor	1986 Amendments	1.30E-04	0.4			5.20E-05	
Heptachlor epoxide	1986 Amendments	2.60E-04	0.2			5.20E-05	
Hexachlorobenzene	1986 Amendments	4.60E-05	1			4.60E-05	
Pentachlorophenol	1986 Amendments	3.00E-06	1	0	2.10E-01	3.00E-06	6.30E-07
Toxaphene	NPDWR	3.20E-05	3			9.60E-05	
1,1,2-Trichloroethane	1986 Amendments	1.60E-06	200			3.20E-04	
Vinyl chloride[3]	1986 Amendments	2.10E-05	2	0	1.80E-01	4.20E-05	3.78E-06
		4.20E-05	2	0	1.80E-01	8.40E-05	7.56E-06

[1] National extrapolation based on sample occurrence data, a

[2] Only one detection each. 95th percentile is taken as this value

[3] Two possible dose-response relationships proposed by EPA

Table 5.6: Risk data for carcinogenic

Carcinogenic contaminant evaluated under the CCL	Population exposed to levels above the 95th percentile			95th-percentile risk		
	Round 1 data	Round 2 data	UCMR 1 data	Round 1 data	Round 2 data	UCMR 1 data
1,3-dichloropropene	100800	58550	NA	4.32E-06	7.60E-05	NA
1,1,2,2-tetrachloroethane	229600	238050	NA	3.30E-04	6.67E-06	NA
2,4-dinitrotoluene[2]	38000	NA	NA	5.55E-03	NA	NA
2,6-dinitrotoluene	0	NA	NA	0	NA	NA
Aldrin	NA	52600	NA	NA	1.48E-03	NA
DDE[2]	NA	NA	17670	NA	NA	2.55E-05
Dieldrin	NA	39650	NA	NA	1.41E-03	NA
Hexachlorobutadiene	95450	251350	NA	7.79E-06	1.26E-06	NA

Regulated carcinogenic contaminant
Arsenic
Arsenic
Acrylamide
Epichlorohydrin
Benzene[3]
Benzo(a)pyrene
Bromate
Carbon tetrachloride
Chlordane
1,2-Dichloroethane
Dichloromethane
Di(2-ethylhexyl) adipate
Di(2-ethylhexyl)phthalate
Dioxin (2,3,7,8-TCDD)
Ethylene dibromide
Heptachlor
Heptachlor epoxide
Hexachlorobenzene
Pentachlorophenol
Toxaphene
1,1,2-Trichloroethane
Vinyl chloride[3]

[1] National extrapolation based on sa

[2] Only one detection each. 95th perc

[3] Two possible dose-response relatio