Springer Water

Mohammed H. Dore

Global Drinking Water Management and Conservation

Optimal Decision-Making

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For my grandchildren: Aidan, Norah, and Liam. May they inherit a clean environment and clean water

Preface

This writing project began as a book on a number of issues affecting drinking water and governmental policy on water resource management. But the range and depth of the material on the subject necessitated that it be split into two companion books, each of which could be read and appreciated independently of the other. As the title of this book indicates, the focus of this book is on a number of theoretical principles that should guide water resource management and drinking water production, both in the developed and developing countries. It makes sense to bring these theoretical principles under one cover, especially this year, as this is the United Nations "International Decade for Action, Water for Life, 2005–2015." The companion book is focused on water policy in Canada. However, each book can be read independently of the other.

In a series of books and reports, Dr. Peter Gleick, President of the Pacific Institute, has carried out painstaking research on a large number of issues relevant to the sustainable use of water resources. His latest biannual report was released in January 2014. This book complements that research with a focus on the management of drinking water, although that cannot be divorced from sustainable water resource management for ecosystem health, the overarching philosophy for sustainable use that German water and other European authorities have explicitly recognized. Maintenance and restoration of ecosystem functioning and health ought now to be recognized as being synonymous with the "social good." But the growing evidence of environmental damage all over the globe makes it clear that the social good is being very narrowly defined. The environmental damage can be seen in stresses on land, air, oceans, and freshwater.

Global freshwater resources are coming under increasing stress, not only due to economic development of middle income and poorer countries but also due to shifting patterns of precipitation due to climate change, whereby the northern hemisphere is getting wetter but some pockets of drier areas getting even drier, such as the mid-southwest of the United States and the drier areas of western Canada. On the other hand in Africa, desertification is advancing and flow rates in the existing rivers and lakes are becoming more variable. Areas in southern Europe can also expect increasing water stress. Under these conditions, conservation of water has increased in importance. Some water-stressed areas are beginning to look for inter-basin water transfers but these are unsound from the perspective of ecosystem health. There is also growing evidence of water conflicts becoming more prominent. A large trade in drinking water in the form of bottled water exists but there is also a search for bulk water exports. For example much of Canada's water flows north, but from time to time there are fears of the possibility of bulk water export or diversion of freshwater from the northern rivers and the Great Lakes into the Mississippi River though the Chicago Diversion for the growing population of the US "sunbelt." Similarly, Turkey has proposed bulk water exports to Israel. Some inter-basin transfers, such as those from the Great Lakes to the south of the US have the potential for future conflict.

Inter-basin water transfers and the potential for conflict can be avoided if there is in place a committed policy of water conservation in order to ensure that ecosystem health is ranked as a priority in water resource management all over the globe. This primary aim needs to be supplemented with systemic adaptation to the changing availability of freshwater through climate change and its effects on the distribution of water. However, rapid (though uneven) economic development is making water scarcity a major threat. As fresh and clean water supply comes under stress, most drinking water is no longer pristine and must be treated for pathogens and other contaminants. In North America, the treatment method is to rely largely on chlorine, primarily to kill bacteria and viruses. But the threats from protozoa remain, and these have led to a number of waterborne disease outbreaks, as chlorine is ineffective against a number of pathogens, as this books shows.

The production of drinking water requires adequate management, with appropriate pricing and management under risk, an idea that the World Health Organization has been promoting in order to reduce or eliminate waterborne disease outbreaks. In this book, the major theoretical issues in the management of drinking water are considered in some detail. These issues are: (1) watershed protection from harmful human industrial, mining and agricultural activity; (2) characteristics of drinking water treatment technologies and their unit prices under conditions of economies of scale; (3) theory and practice of water pricing; (4) methods and processes of adopting risk assessment in drinking water management; (5) up-to-date water infrastructure management incorporating risk; (6) a serious commitment to overcome risks to long-term health through reduced reliance on chlorine and chlorine derivatives for disinfection; (7) an inadequate response to the threat of lead in drinking water; and (8) poor management of wastewater that becomes the source of drinking water, with the concomitant presence of micro-pollutants in the drinking water. All this is the subject of this volume. In a companion book, the focus is government-level policy on water in Canada. As water is a provincial responsibility, there are separate chapters on water policy in four provinces: Ontario, Alberta, British Columbia, and Newfoundland and Labrador.

Returning to this book, and the key principles, a word about how water supply is organized in some developed countries. Some large cities in Europe operate water supply as a private but regulated business. However, in much of the world water is almost exclusively provided by a local municipality, as a local "public" good.

Naturally in this case there is no profit motive, and no incentive to innovate, introduce more advanced technology, and to improve water quality. The European private companies and other pockets of privatized water companies seem well managed, but it is not clear that they are innovators in delivering higher water quality. What seems to lead to higher quality drinking water is government leadership through adequate regulation, as in Denmark, the Netherlands, and Germany. When the public becomes aware of what is possible and finds out what has been done in other jurisdictions, such as Denmark, the Netherlands, and Germany, then perhaps public awareness will push their own governments and their utilities to improve water quality.

There are two long-term threats to health associated with the treatment and delivery of drinking water: one is the presence of lead in drinking water, which is a serious health hazard. It is therefore imperative that the lead content of drinking water is properly measured; there are two chapters that deal with lead in drinking water (Chaps. 10 and 11). The other long-term threat is the use of chlorine and chlorine derivatives used in the disinfection of drinking water (Chap. 9). The use of chlorine results in a large number of "disinfection byproducts," some of which are regulated in the developed countries. But chlorine alone is ineffective against protozoa, and the byproducts carry some very long-term threats to human health. There are new treatment technologies that do not have these byproducts and are therefore safer. These newer technologies can be used to deliver a higher quality of water, but there appears to be lack of knowledge of these possibilities, and possibly apathy among governments. Consumers might demand better water quality if they had more information on the new technologies and their costs.

Communities in Europe seem more cognizant of some of the long-term threats to health associated with the use of chlorine as a primary disinfectant, but other threats due to lead in the water remain a major concern, although there are some European countries (like Denmark) where this threat is taken very seriously and largely eliminated. But in the rest of the world the presence of lead in old pipes and even in the treatment systems continues to be a concern. For the threat of lead, what is required is a chemically sound lead sampling protocol and an appropriate maximum contamination level (MCL) set as a regulation. It would also help if there was a systematic plan to eliminate all lead pipes and fixtures.

Most developed countries have strong regulations against the presence of pathogens and once lead is eliminated, the next frontier in water quality will be the elimination of chemical contaminants such as *pesticides* (e.g. *atrazine*), *herbicides, pharmaceuticals, and personal care products*. This is a problem when the source water comes from multi-use watersheds like the Great (North American) Lakes. Europe has made more progress; most European jurisdictions have moved away from surface water as a source and switched to groundwater, which by itself is a natural form of "treatment"; groundwater is often free of contaminants except where there are known contaminants, such as iron and manganese.

It could be argued that smaller countries like Denmark and the Netherlands can afford to be aggressive in assuring better quality of water. But the case study of Germany reported in this book shows what can be done to improve drinking water quality by avoiding some of the long-term risks. Germany offers some important lessons both for North America and for the developing world on how water supply could and should be managed.

I hope that the coverage of these important topics in the management and delivery of clean water will stimulate discussion on what can be learnt from Germany to help improve drinking water quality everywhere, including the developing countries. Thus the book is oriented toward filling the knowledge gap and showing the potential for improvement. As such it is likely to be of interest to water system owners, managers, water engineering consultants, and regulators all over the world. The comparative dimension may also appeal to some readers, to see how some jurisdictions manage their water supply as a public service producing a product essential to life.

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I should like to record all the help that I have received in writing this and the companion book. First, the two books would not have been possible without the research grants that I have been fortunate enough to receive from the Social Sciences and Humanities Research Council of Canada (SSHRC), The National Science and Engineering Council of Canada (NSERC), the Canadian Foundation for Climate and Atmospheric Sciences (CFCAS),[1] the US National Science Foundation (US-NSF), the Climate Change Action Fund of the Federal Government of Canada, and grants for teaching release from Brock University, which in turn were possible thanks to the Research Time Release Stipends included in my SSHRC grants over the last few years. The research grants enabled me to establish my Climate Change Lab at Brock University. In this lab I was fortunate in hiring many of my students as research assistants, and most of them wrote their graduate or undergraduate Honors theses under my supervision in the lab. They have greatly influenced my thinking and many contributed important germs of new ideas, and new models as vehicles of inquiry; these dramatically altered my thinking, as teaching is a two-way enriching process. I want to record my debt to all my former students, who are now well established in their own careers. The names that I remember most (in alphabetical order) are: Abba Ansah, Katherine Ball, Geoff Black, Ryan Bruno, Hassan Chilmeran, Ridha Chilmeran, Eric Eastman, Ken Gilmour, Clay Greene, Indra Hardeen, Ryan Harder, Aaron Janzen (at the University of Calgary), Jamie Jiang, Mathew Chang Kit, Ryan Kwan, Soomin (Tomy) Lee, Tony Lipiec, Roelof Makken, Michael Patterson, Jeff Pelletier, Sasha Radulovich, Angela Ragoonath, Noureen Shah, Amar Shangavi, Peter Simcisko, Rajiv Singh, Harvey Stevens, Mireille Trent, and Klemen Zumer. They all cut their "research" teeth in my lab but gave much of their time and effort and are now my friends. While some are completing PhDs, others are well advanced in their professional careers; one of them (Roelof Makken) generously established the "Mohammed Dore Graduate

[1] Now transformed by the Federal Government into the "Canadian Climate Forum," and no longer a granting agency.

Research Scholarship" at Brock University and is now an adjunct Professor at Brock University, where he has taken over some of my teaching. Jamie Jiang in particular has taken on much of the econometric estimation work and as well as the editorial work of these two books. Her work is meticulous and painstaking; she leaves my lab in the Fall of this year to start her Ph.D. program. I think of all of my former students as my co-authors of these two books; I cannot imagine how I would have functioned without them.

My thanks also go to the Deans of the Faculty of Social Sciences (Deans David Siegel and Thomas Dunk) and the Office of the Vice President, Research Services; their help has been invaluable. The chapter on Germany was read by two people in Germany: my good friend Dieter Jablonka and Mr. Michael Schneemann, water engineer at *Wasserbeschaffungsverband*, the water utility in Harburg, Germany. Mr. Schneemann's comments and suggestions were very helpful. I also received help and advice from Prof. Dr.-Ing. Helmut Grüning, at the IWARU Institute of Water in Münster and from Dr. Christiane Markard, Head of Division II, "Environmental Health and Protection of Ecosystems," at *Umweltbundesamt*, which is the Environmental Protection Agency of the Federal Republic of Germany. But I alone am responsible for the contents of this book and for any remaining deficiencies.

I must thank Margaret Dore who over the years has read and edited *all* my books and many of my articles. She has read and improved many successive drafts of the two books being published by Springer. Finally I wish to record my thanks to my Editor, Dr. Tobias Wassermann, at Springer for constructive comments and constant encouragement; in many ways he is an ideal editor.

July, 2014

Contents

Part I
Waterborne Diseases and Watershed Protection

In Part I, we seek answers to the following questions:

- Are there any patterns in the nature of distribution of waterborne disease outbreak?
- What pathogens cause the most serious outbreaks?
- What are the causes of these outbreaks?
- What lessons can be learned from these outbreaks?

Chapter 1
Introduction to Drinking Water Management

1.1 An Apologia or Why I Wrote This Book

Why should an economist write a book on water resource and drinking water management? What are the principles of economic theory that are relevant to the desirable objective of clean and healthy water, not only for human consumption but also for ecosystem health?

One obvious answer is the presence of what economists call "externalities." There is no doubt that modern agricultural, mining, and industrial activity has indeed raised the wellbeing of citizens, but this has come at a certain *social* cost that is not taken into account. A negative externality is nothing more than an unaccounted social cost. Economics argues that that is precisely when the State must intervene "in the interests of society and future generations." When the State fails to take adequate corrective action, we see evidence of environmental degradation. It is this failure of adequate control, regulation, and management of not only treated drinking water but also the *sources* of drinking water that we see in many parts of the world, including North America. The motivation behind this book is also provided, in part, by the fact that we have a sorry record of waterborne disease outbreaks that clearly carry the message that not all is well in the way we care for drinking water, pay for it, and then dispose of the wastewater into the very watercourses that are our drinking water sources. The failure of an adequate government response to deal with the externalities is also indicative of the decline in the role of government from what might be called optimal from a social point of view. It is sometimes forgotten that the guiding principle of economics is the implementation of the "social good," although this "social good" is typically interpreted as a "competitive equilibrium," in which no negative externalities exist, or have been "corrected" by appropriate State action.

Note that this social good does not *necessarily* involve redistribution of income to enhance wellbeing of some, or invoke the *Rawlsian difference principle* for a "liberal" society (Rawls 1971). That of course requires an activist State. Our

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argument is based on the minimal libertarian grounds on which economic theory relies; it mandates state action to “correct” or ameliorate a violation of property rights, such as a misuse of public property to the disadvantage of current and future generations (Dore 1998). Since economic theory assumes the need for property rights and mechanisms to enforce those rights, a minimal “night watchman state,” of the sort proposed by Nozick (1974), may be assumed in standard neoclassical economic theory. The so-called Coasian approach of “let-them-negotiate” to deal with externalities (Coase 1960) is a legalistic accretion into economics, made respectable when Ronald Coase was awarded the Bank of Sweden Prize (in memory of Nobel) in 1991. This bilateral Coasian negotiation is not possible when the injured part is “society” or future generations and hence the Coase “theorem” is not applicable. In contrast to Coase, strict economic theory has a legitimate set of tools for rectifying negative externalities, from corrective taxes to controls. But what economic theory is powerless to do is to provide the *political will* to enforce a “socially correct” intervention or solution.

Indeed we could go further: the developments in the new public economics that arose after the seminal contributions of Professor Sir James Mirrlees in the early 1970s and what has followed since, show that the instruments that were previously thought to be economically “illegitimate” (like quantity controls, quotas, forced savings plans, prohibitions, etc.) can be seen as social “improvements,” necessitated and indeed *justified* in an economy that is already distorted by a whole lot of nonlinearities situating it far away from a hypothetical competitive equilibrium (see more on this in Chap. 5).

I conclude that there is absolutely no reason why an economist, armed with such a robust body of thought and conceiving economics to be a social and moral science that is dedicated to the betterment of social life, might not legitimately write about water resources and drinking water management. In fact it is only with such a *social* perspective that the findings of the sciences of hydrology, limnology, epidemiology and bio-eco-system functioning can be utilized for the preservation of the biome in this *anthropocene* age, an age characterized by the adverse and negative impacts of human activity. Hence, no further justification for writing this book is necessary; very few scientists and even economists would be surprised that a whole array of economic concepts and econometric and statistical tools can be used to carry out a concerted critique of current management and social policy with the objective of improving current policy and practice. Carrying out such a critique is one of the objectives of this book as well as a companion book, which is focused on a critical appraisal of water policy in Canada.

1.2 Water in a Global Context

Between 2009 and 2050, the world population is expected to increase from 6.8 to 9.1 billion (UN-DESA 2009). At the same time, urban populations are projected to increase by 2.9 billion, from 3.4 billion in 2009 to 6.3 billion total in 2050. So most

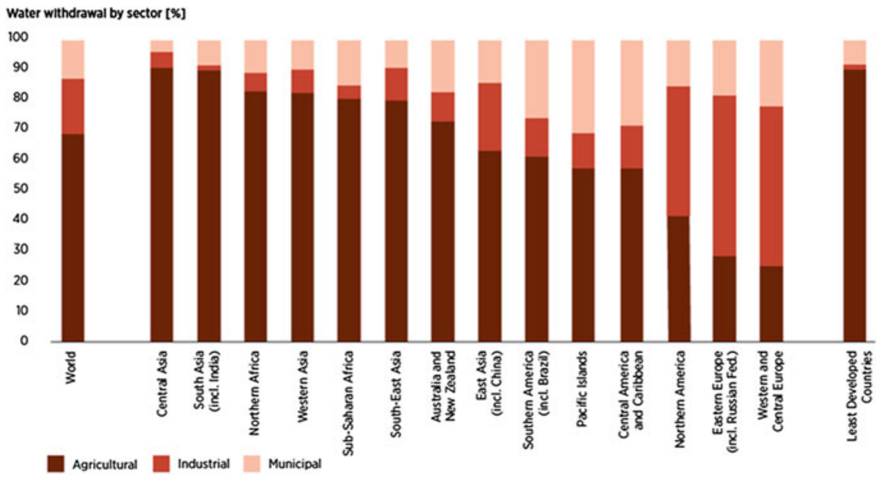

Fig. 1.1 Water withdrawal by sector by region in 2005 (WWAP 2012)

of the growth in population is likely to be in urban areas of the world (UN-HABITAT 2006). Worldwide, 87 percent of the population gets its drinking water from treated sources, and the corresponding figure for developing regions is also high at 84 percent. However, access to clean water is far greater in urban areas (at 94 percent), while only 76 percent of rural populations have access to treated water (WHO/UNICEF 2010).

Water for irrigation and food production constitutes one of the greatest pressures on freshwater resources. Agriculture accounts for about 70 percent of global freshwater withdrawals; the sectoral distribution amongst major country groupings is shown in Fig. 1.1. Global population growth, combined with changing diets, is predicted to increase food demand by 70 percent by 2050. Clearly this has implications for water demand as well.

Groundwater abstraction in 2010 was estimated to be around 1,000 km^3, of which two-thirds was for irrigation and the rest divided between industrial and domestic uses (see Fig. 1.1 again). Estimates suggest that groundwater abstraction represents 26 percent of total global water withdrawal but the global groundwater recharge rate is only 8 percent. Total stored groundwater is poorly known; estimates range from 15 to 60 million km^3, including 8–10 million km^3 of freshwater, while the remainder (brackish and saline groundwater) is found mainly at great depth (Margat 2008). There is some evidence that significant groundwater storage depletion is taking place in many areas of the world.

Globally, desertification, land degradation, and drought affect 1.5 billion people who depend on degraded lands. Some 42 percent of the very poor live on degraded lands, compared with 32 percent of the moderately poor and 15 percent of the non-poor (Nachtergaele et al. 2010). India alone accounts for 26 percent of the population affected by desertification and drought; China 17 percent, and sub-Saharan Africa 24 percent; the remaining part of Asia-Pacific 18.3 percent; Latin America and the Caribbean 6.2 percent; and north east and north Africa 4.6 percent (ICRISAT 2008). Desertification and droughts have their greatest impact in Africa where two-thirds of the continent is desert or is water scarce.

In economic theory, drinking water is partly a public good and partly a normal consumption good. In North America, a large portion of drinking water is used outside the home as a normal good for uses such as gardening and washing cars. Domestic water use in the US averages between 80 and 100 US gallons (302–378 L), whereas in Canada the per capita consumption is 343 L, but only about 10 percent of it is for drinking and cooking. However, with very few exceptions, all publicly supplied water is treated to drinking water standard, partly for fear that untreated water could lead to illness.

1.2.1 Climate Change and Water

Dore (2005) surveyed the evidence for changing global patterns of precipitation. This subsection is based on those findings. It appears that annual land precipitation has continued to increase in the middle and high latitudes of the Northern Hemisphere (very likely to be 0.5–1 percent per decade), except over Eastern Asia. Over the subtropics (10° N–30° N), land-surface rainfall has decreased on average (likely to be about 0.3 percent per decade), although this has shown signs of some recovery. But this recovery could simply be evidence of increased variability. Tropical land-surface precipitation measurements indicate that precipitation has probably increased by about 0.2–0.3 percent per decade over the twentieth century, but increases are not evident over the past few decades and the amount of tropical land (versus ocean) area for the latitudes 10° N–10° S is relatively small. Nonetheless, direct measurements of precipitation and model reanalyses of inferred precipitation indicate that rainfall has also increased over large parts of the tropical oceans. Where and when available, changes in annual stream-flow often relate well to changes in total precipitation. The increases in precipitation over the Northern Hemisphere mid- and high-latitude land areas have a strong correlation to long-term increases in total cloud amount. In contrast to the Northern Hemisphere, no comparable systematic changes in precipitation have been detected in broad latitudinal averages over the Southern Hemisphere.

Decreasing snow-cover and land-ice extent are positively correlated with increasing land-surface temperatures. Satellite data show that there is very likely to have been decreases of about 10 percent in the extent of snow cover since the late 1960s. There is a highly significant correlation between increases in Northern Hemisphere land temperatures and decreases in snow cover. There is ample evidence to support a major retreat of alpine and continental glaciers in response to twentieth-century global warming. This evidence has continued to grow over the period 2010–2014. In a few maritime regions, increases in precipitation due to regional atmospheric circulation changes have overshadowed increases in temperature in the past two decades, but overall glaciers in the northern and southern hemispheres have continued to shrink.

Over the past 100–150 years, ground-based observations show that there is very likely to have been a reduction of about 2 weeks in the annual duration of lake and river ice in the mid- to high latitudes of the Northern Hemisphere. New analyses show that in regions where total precipitation has increased, it is very likely that there have been even more pronounced increases in heavy and extreme precipitation events. The converse is also true. In some regions, however, heavy and extreme events (i.e. defined to be within the upper or lower 10 percentiles) have increased despite the fact that total precipitation has decreased or remained constant. Where this has occurred, it is attributed to a decrease in the frequency of precipitation events. Overall, it is likely that for many mid- and high-latitude areas, primarily in the Northern Hemisphere, statistically significant increases have occurred in the proportion of total annual precipitation derived from heavy and extreme precipitation events; it is likely that there has been a 2–4 percent increase in the frequency of heavy precipitation events over the latter half of the twentieth century. For the Southern Hemisphere, there is some concern that while extreme precipitation events have increased, total annual precipitation may have declined (Dore and Singh 2013; Dore and Simcisko 2013).

Over the twentieth century (1900–1995), there were relatively small increases in global land areas experiencing severe drought or severe wet conditions. In some regions, such as parts of Asia and Africa, the frequency and intensity of drought have been observed to increase in recent decades. In many regions, these changes are dominated by inter-decadal and multi-decadal climate variability, such as the shift in the El Niño Southern Oscillation (ENSO) toward more warm events. But there is great uncertainty over the change in the frequency and variability of El Niño and La Niña events, which typically have a global influence on the distribution of precipitation. Ocean currents continue to be major influences on precipitation everywhere on the globe and so possible changes in any of the major ocean currents could change precipitation drastically.

Other statistical analyses of rainfall patterns in some of the dryland regions reveal a steep drop in the early 1970s, which has persisted, a reduction of about 20 percent in precipitation levels resulting in a 40 percent reduction in surface runoff (EU, Council of the European Union 2007). Furthermore, the International Water Management Institute predicts that climate change will have dire consequences for feeding an ever-expanding global population, especially in areas of Africa and Asia where millions of farmers rely solely on rainwater for their crops. In Asia, 66 percent of cropland is rain-fed, while 94 percent of farmland in sub-Saharan Africa relies on rain alone, according to the International Water Management Institute (IWMI 2007). These are the regions where water storage infrastructure is least developed and where nearly 500 million people are at risk of food shortages.

There is no doubt that the changing pattern in the observed precipitation is the signature of global climate change. That is, precipitation is being globally reallocated by climate change. Perhaps it is the least developed that will experience the most adverse consequences of climate change. Richer countries have now lived with Third World poverty for decades and will view more disasters there, aggravated by extremes of climate, as nothing new. The consequences of global warming are more likely to be treated as calling for voluntary acts of charity than as a matter of equity, requiring compensation for the actions of the industrialized countries. That will be the greatest inequity of global climate change. The patterns sketched above have now been confirmed with even greater confidence by the IPCC Fifth Assessment Report (IPCC 2014).

The above section is a brief outline of "the state of the biome"; the adverse consequences of human actions coupled with advances in medicine and economic development are likely to have contradictory impacts on the world. Perhaps the most serious threat over the next 50–100 years will be the impacts of climate change, and the most severe impacts are likely to be on water resources: dry areas getting even drier and wet areas enduring more precipitation, with more extremely heavy precipitation causing flooding, property damage, and loss of life. It is this rather precarious context within which human societies will have to manage the provision of safe drinking water.

1.3 What This Book Is About

This book is concerned with the comparative management of drinking water in the developed, richer countries, who in principle have the resources to give their citizens the best and highest quality drinking water and yet so often fail to do so. The management of water in the developing countries is an even more daunting task, as they do not have the financial resources or the knowledge of treatment technologies. Both in the developed and the developing world, the crisis is partly due to lack of public funding for small and rural communities, partly due to government complacency, but also due to lack of knowledge. For example, some jurisdictions (such as Alberta, and Newfoundland and Labrador in Canada, and parts of Europe) are more proactive and innovative in capital support and in the adoption of new technology; some communities are prepared to pay a higher price for water when water is privatized, as in some countries in Europe. But there is a serious knowledge gap about (a) water treatment technologies and their costs, (b) risk assessment methods, (c) adverse health effects of chemical contaminants, (d) management protocols, and (e) varying regulatory practices in different jurisdictions, and what successes are possible even with small financial outlays. This book is about these issues. It begins with a record of waterborne disease outbreaks, and the lessons learned from that. That lesson is the need for a multi-barrier approach to the protection of drinking water. The first component of the multi-barrier approach is adequate watershed protection. The book then proceeds with a comparative classification of water treatment technologies. The classification is based on the *contaminants removed;* this is an indirect way to get to "water quality," which also depends on the quality of source water in the first place. By focusing on the contaminants removed, we get a sense of the water quality associated with any given treatment technology.

It is also obvious that drinking water can be made safer if watershed contamination from human activities is minimized; these principles of watershed management are well known in the literature and are summarized briefly in Chap. 2, and explained in detail in Chap. 6. Furthermore, a water utility can improve water quality by better management of its infrastructure for the benefit of the public. This is less well known, and so two chapters are devoted to infrastructure asset management that incorporates risk (Chaps. 7 and 8).

Some large cities in Europe operate water supply as a private but regulated business. However, in much of the world water is almost exclusively provided by a local municipality, as a local “public” good. Naturally in this case there is no profit motive, and no incentive to innovate, use more advanced technology, and improve water quality. The European private companies and other pockets of privatized water companies seem well managed, but it is not clear that they are innovators in delivering higher water quality. What seems to lead to higher quality drinking water is government leadership through adequate regulation, as in Denmark, the Netherlands, and Germany (Chaps. 9–12). When the public becomes aware of what is possible and finds out what has been done in other jurisdictions, such as Denmark, the Netherlands, and Germany, then perhaps public awareness will push their local governments and their utilities to improve water quality.

As shown in Chap. 3, the production of drinking water is characterized by strong economies of scale, which give large cities a cost advantage and all small and rural communities (the majority of water systems) a serious disadvantage. This affects the choice of water treatment technology for drinking water. Some jurisdictions recognize this factor and compensate for it through special programs, while others let the small communities fend for themselves. This creates an asymmetry, with small communities meeting the minimum regulatory requirements, with periodic crises, while the larger cities receive water with a lower probability of disease outbreaks. However, in all communities that merely meet the minimum regulatory requirements, long-term threats to health are often ignored. There are two long-term threats to health associated with the treatment and delivery of drinking water: one is the presence of lead in drinking water, which is a serious health hazard. It is therefore imperative that the lead content of drinking water is properly measured; there are two chapters that deal with lead in drinking water (Chaps. 10 and 11). The other long-term threat is the use of chlorine and chlorine derivatives used in the disinfection of drinking water (Chap. 9). The use of chlorine results in a large number of “disinfection byproducts,” some of which are regulated in the developed countries. But chlorine alone is ineffective against protozoa, and the byproducts carry some very long-term threats to human health. There are new treatment technologies that do not have these byproducts and are therefore safer. These newer technologies can be used to deliver a higher quality of water, but there appears to be a lack of knowledge of these possibilities, and possibly apathy among governments. Consumers might demand better water quality if they had more information on the new technologies and their costs.

Communities in Europe seem more cognizant of some of the long-term threats to health associated with the use of chlorine as a primary disinfectant, but other threats due to lead in the water remain a major concern, although there are some European countries (like Denmark) where this threat is taken very seriously and largely eliminated. But in the rest of the world the presence of lead in old pipes and even in the treatment systems continues to be a concern. For the threat of lead, what is required is a chemically sound lead sampling protocol and an appropriate maximum contamination level (MCL) set as a regulation. It would also help if there was a systematic plan to eliminate all lead pipes and fixtures.

Most developed countries have strong regulations against the presence of pathogens and once lead is eliminated, the next frontier in water quality will be the elimination of chemical contaminants such as *pesticides* (e.g. *atrazine*), *herbicides*, *pharmaceuticals, and personal care products*. This is a problem when the source water comes from multi-use watersheds like the Great (North American) Lakes. Europe has made more progress; most European jurisdictions have moved away from surface water as a source and switched to groundwater, which by itself is a natural form of “treatment”; groundwater is often free of contaminants except where there are known contaminants, such as iron and manganese.

It could be argued that smaller countries like Denmark and the Netherlands can afford to be aggressive in assuring better quality of water. For that reason we have chosen Germany as a case study of what can be done to improve drinking water quality by avoiding some of the long-term risks. Germany has a population of 82.6 million (in 2014). It offers some important lessons both for North America and for the developing world on how water supply could and should be managed.

I hope that the coverage of these important topics in the delivery of clean water will stimulate discussion on what can be learned from Germany to help improve drinking water quality everywhere, including the developing countries. Thus, the book is oriented toward filling the knowledge gap and showing the potential for improvement. As such it is likely to be of interest to water system owners, managers, water engineering consultants, and regulators all over the world. The comparative dimension may also appeal to some readers, to see how some jurisdictions manage their water supply as a public service producing a product essential to life.

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Chapter 2
Waterborne Disease Outbreaks and the Multi-barrier Approach to Protecting Drinking Water

2.1 Introduction

Drinking water outbreaks have occurred throughout the world, causing varying illnesses and even death. This chapter reviews past outbreaks of microbial contaminants and the associated lessons that have been learnt. Only the most recent outbreaks are considered, as these are probably the most relevant for policy purposes. The publication, *Safe Drinking Water: lessons from recent outbreaks in affluent nations,* written by Hrudey and Hrudey (2004), summarize the occurrences of 69 drinking water outbreaks. They begin in January 1974 in Richmond Heights (Florida), and continue up to March 2002 in Transtrand (Sweden). The Hrudeys have made a significant contribution to the topic of drinking water safety through the detailed account of these outbreaks, and their overall analyses. Their book emphasizes the impact of the Walkerton contamination of 2000, and also describes the lessons that have been gained as a result. While taking the Hrudeys' work into consideration, the objective of this chapter is to include outbreaks that have occurred since 2002, and to expand upon their conclusions.

A variety of contaminants have caused water outbreaks, but a few in particular are a primary concern. *Cryptosporidium parvum*, *Giardia lamblia*, *Campylobacter jejuni*, and *Escherichia coli* (*E. coli*) have caused the largest and most significant outbreaks, and therefore are the main focus. *Toxoplasma gondii* is also included, but this pathogen is very rare and does not pose the same level of risk. These disease-causing contaminants are most commonly transmitted into water sources by animal or human fecal matter. The three categories of microbial contaminants in drinking water are protozoa, bacteria, and viruses. Protozoa and bacteria contaminants have had the most significant impacts and are the focus of this review.

Outbreaks can occur because of an array of factors. The multi-barrier approach is the primary strategy for enhancing safety of drinking water systems. This approach is designed to provide the best quality of water by using a number of checkpoints throughout a water system. If a contaminant enters the water system, it is the goal of

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the multi-barrier approach to detect and treat the contaminant before it reaches the consumer. A failure in one or more of the barriers can lead to the spread of contamination, which results in an outbreak if not detected in the remainder of the system. Most outbreaks in the history of drinking water contamination have been a result of barrier failures. Failures can occur anywhere within the system, including source water, operations, treatment, and distribution. Outbreaks are an indicator of weaknesses within a water system. The number of outbreaks and their severity reflect poorly on the ability of an overall system to provide safe drinking water. Also, the weather frequently plays a key role in outbreaks by introducing contaminants into water sources, often by runoff from either a heavy rainfall or spring melt. Surface water is particularly vulnerable to weather occurrences because it is easily accessible, in contrast to groundwater sources that have natural filtration through the soil, and therefore incur less contamination than surface water, in general. Over time, with the knowledge gained through the experience of past outbreaks, fewer outbreaks would be expected to occur as water systems improve to prevent future contamination. However, this has not been the case, as contamination continues to be a major concern throughout the world, and surprisingly even in developed countries such as Canada, the United States, and Europe.

This chapter is organized as follows: Protozoa contaminants of *Cryptosporidium*, *Giardia*, and *Toxoplasma* are discussed first. This is followed by a discussion of bacterial contaminants of *Campylobacter*, and *E. coli*. For each contaminant a description is included as well as the predominant outbreaks that each has caused. Furthermore, a number of principles of watershed management are reviewed in Sect. 5.

2.2 Protozoa

2.2.1 Cryptosporidium

Cryptosporidium is a frequent microbial cause of drinking water outbreaks. This is because as a protozoan it is resistant to chlorine disinfection, which allows it to spread undetected throughout the distribution system to the consumer. Chemical disinfection is a critical barrier in the multi-barrier approach to prevent the possible spread of contamination, but alternative treatment is necessary for systems to be effective against *Cryptosporidium*. Alternative treatments including coagulation, sedimentation, filtration, ozone and ultra-violet light treatment have been determined to be effective against *Cryptosporidium* (Rose 1997, p. 149). Communities that rely solely on chlorine are the most vulnerable to an outbreak of cryptosporidiosis, the disease caused by *Cryptosporidium*. Severe diarrhoea is the main symptom of cryptosporidiosis. The majority of the outbreaks are related to treatment failures within the water system, often a heavy reliance on chlorine and a failure to provide filtration. Filtration is extremely important in removing *Cryptosporidium*, particularly in communities that rely on surface water sources. This is because of the vulnerability of surface water to potential contaminants in the surrounding environment. Therefore, cryptosporidiosis occurs most commonly in communities that rely on surface water and do not provide filtration.

A well-documented outbreak of cryptosporidiosis occurred in Braun Station, Texas in May–July of 1984. The groundwater source was chlorinated, but was believed to have been contaminated by sewage (Rose 1997, p. 141). The outbreak caused 2,000 cases of reported illness in the community of approximately 5,900 people. A lack of effective treatment was the system failure that caused this outbreak. *Cryptosporidium* is resistant to chlorine and the treatment facility did not provide filtration, which would have been effective against the pathogen.

In January–February of 1987 Carrollton Georgia experienced an outbreak of cryptosporidiosis. The surface water source became contaminated and caused over 13,000 cases of illness. At the time of the outbreak approximately 27,000 people were supplied by the water system in the county of 64,900 people. The source of contamination is believed to have been fecal runoff from nearby grazing cattle and sewage overflow from upstream into the river source (Solo-Gabriele and Neumeister 1996, p. 79). Conventional treatment was used prior to the outbreak, which includes coagulation, flocculation, sedimentation, filtration, and chlorination. Improper flocculation, which is part of the filtration process, allowed the contaminant to spread through the distribution system (Rose 1997, p. 141). This means that the system failure in this outbreak is inadequate filtration. This is a failure in the treatment process of the water system, and therefore could have been prevented by a properly working system. Of course it could be also argued that there was inadequate monitoring since the operators did not detect, and therefore did not fix the improperly working flocculation. Following the outbreak, the treatment system was upgraded in Carrollton with new flocculators, increased filter monitoring, improved chemical dosage, and operational practices (Solo-Gabriele and Neumeister 1996, p. 81). These alterations are significant improvements in an effort to prevent future contamination.

In January–June of 1992, Jackson County Oregon also experienced an outbreak of cryptosporidiosis. The contamination of the surface water sources caused 3,000 cases of defined illness, but is estimated to have affected approximately 15,000 people. The outbreak occurred within two water supplies of Jackson County. The water system of the city of Medford supplied 70,000 people, and the city of Talent's system supplied 3,000 people (Solo-Gabriele and Neumeister 1996, p. 79). The water in Medford came from Big Butte Springs, and was treated with chlorine only. In Talent the water came from a river source that was treated with flocculation, sedimentation, filtration, and chlorination. The source of contamination of Medford's springs is believed to have been contaminated surface water and the source of Talent's river contamination is believed to have been treated wastewater that was received by the river. Drought conditions at the time may have lessened its dilution in the river (Solo-Gabriele and Neumeister 1996, p. 80). Another suggested source for Talent is agricultural runoff that possibly could have entered the river through runoff of rainfall. Again system failures were involved in Medford's outbreak with a lack of filtration and a sole reliance on chlorination, and also in Talent with poor filtration that did not take into account increased in turbidity at the time (Rose 1997, p. 141). Following the outbreak, Medford flushed its distribution system with chlorinated and filtered river water, and Talent initiated corrections to their system deficiencies such as equipment repairs and treatment alterations (Solo-Gabriele and Neumeister 1996, p. 81).

In January–June of 1992 North Cumbria in England experienced cryptosporidiosis, with an undetermined number of cases. A third of the population of 160,000 was supplied with water from Ennerdale Lake, another third was supplied by Crummock Lake, and the remaining third was supplied by smaller sources (Goh et al. 2004, p. 1007). The contamination occurred in Ennerdale Lake and the source is believed to have been runoff from nearby livestock. The treatment of Ennerdale Lake entailed only chlorination, and therefore a lack of filtration was the system failure that allowed the outbreak to spread in North Cumbria.

From November 1992 to February 1993, Warrington England experienced an outbreak of cryptosporidiosis. There were 47 confirmed cases and an estimate of approximately 1,840 people affected by the contaminated water that was supplied to 38,000 people (Bridgman et al. 1995, p. 557). Only chlorine was used to treat the groundwater supply. No filtration was applied, which is common for groundwater sources because of the natural filtration achieved through the soil. The source of this outbreak is believed to have been agricultural runoff. It is rare to experience *Cryptosporidium* contamination in a groundwater supply, but it is believed that heavy rainfall caused surface water, contaminated from a field with livestock fecal matter, to drain into the groundwater. Research suggests that the use of groundwater sources establishes a low immunity to *Cryptosporidium* so that a contamination will create a more severe outbreak than may occur in communities that have higher immunities from the use of surface water (Frost et al. 1997, p. 10). The system failure in this case was determined to be a lack of monitoring of the water supply. This is likely because of the rarity of *Cryptosporidium* in groundwater sources, and therefore monitoring of the pathogen was not a regular practice.

The most significant drinking water outbreak of cryptosporidiosis was in Milwaukee Wisconsin from March to April of 1993. Two water treatment plants supplying water to Milwaukee used water from Lake Michigan. The southern plant, that supplied southern and central Milwaukee, became contaminated in April 1993. This caused its temporary closure until June 1993. During this period the northern plant was required to supply the entire area (Osewe et al. 1996, p. 298). Both plants use conventional treatment of coagulation, flocculation, sedimentation, rapid sand filtration and chlorination treatment (Solo-Gabriele and Neumeister 1996, p. 81). The outbreak caused over 403,000 cases of illness and 100 deaths out of approximately 840,000 people whom the water system supplied at the time. The source of contamination is believed to have been from cattle runoff and also human sewage that was carried by tributary rivers (Solo-Gabriele and Neumeister 1996, p. 78). Another suggestion, in other research, is that the source was recycled backwash waters (Rose 1997, p. 141). The recycling of backwash waters is commonly practiced to clean filters by flowing water through in the opposite direction to remove captured particles of matter, in efforts to conserve water (Rose 1997, p. 142). The Milwaukee outbreak has been the largest reported outbreak, and its significance caused changes within the regulation of drinking water in the United States. After the outbreak, the US Environmental Protection Agency enacted the Surface Water Treatment Rule (SWTR). The rule required both disinfection and filtration of all surface waters, as well as groundwaters that are affected by surface waters (Rose 1997, p. 154). Also following the outbreak stricter practices were imposed for chemical dosing and filter monitoring, and long-term improvements were achieved by installing an ozone disinfection facility. Therefore, it can be assumed that the level of filtration and its monitoring must have been inadequate, as it should have been effective at removing or inactivating the contaminant. The contaminants were able to enter the surface source and were also able to pass through the treatment process into the distribution system without detection.

In March 1993, Kitchener/Waterloo Ontario experienced an outbreak of 1,000 cases of cryptosporidiosis. The contamination occurred when the region of approximately 390,000 people switched from a *Cryptosporidium*-free groundwater source to a contaminated surface water source, the Grand River (Frost et al. 1997, p. 10). A newly constructed filtration plant was being used for the conventional treatment of the surface water, with also ozonation treatment. Several other communities had been using the river as a source of water for a number of years, and had not experienced an outbreak. It has been suggested in the literature that this may be related to the low immunity to *Cryptosporidium* that occurs from drinking from groundwater sources (Frost et al. 1997, p. 10). The source of the contamination is believed to have been recycled backwash waters, as suspected in Milwaukee (Rose 1997, p. 141). In the presence of a contaminant, cleaning with backwash water may reintroduce the pathogen into the system. The significance of this outbreak is that it occurred in a large municipality, compared to the majority of drinking water outbreaks that occur in small rural communities.

In 1996, two communities in British Columbia experienced outbreaks of cryptosporidiosis. The first occurred in May in the city of Cranbrook, which has a population of 18,131, causing approximately 2,000 cases of illness. The second occurred shortly afterwards in June in the city of Kelowna, which has a population of 89,442, causing 10,000–15,000 cases of illness. Cranbrook is in the area of southeastern B.C., while Kelowna is in central B.C., 271 km away from Cranbrook (Ong et al. 1999, p. 64). Both cities use surface water sources. Cranbrook uses Joseph Creek and Gold Creek, and Kelowna uses Okanagan Lake. Also, both cities use the same treatment method of only chlorine, without filtration. The majority of water systems in BC are unfiltered and the water is drawn from surface sources, and most also rely on chlorination for simple disinfection (Ong et al. 1999, p. 67). This is a primary concern because of the vulnerability of surface water to contamination. Chlorine is ineffective against protozoan pathogens such as *Cryptosporidium*, and therefore alternative treatment methods, such as filtration, are necessary. The source of contamination in both cases is believed to have been runoff of cattle manure (Ong et al. 1999, p. 63). Treatment failure was the cause of this outbreak because of reliance on chlorine alone. Following the outbreak, the City of Cranbrook decided not to install a filtration plant, but instead has placed monitors into the creeks. Kelowna took more action, possibly because of its larger population and the greater severity of the outbreak, by approving plans for the construction of an ultra-violet light treatment facility. Ultra-violet (UV) treatment has been proven to be effective against protozoa, and therefore is the right step toward preventing future outbreaks.

In May 2000, August 2000, and April 2001 three outbreaks of cryptosporidiosis occurred in Northern Ireland. Respectively, 230, 117, and 129 cases of illnesses were reported in unrelated cases in different locations within the Belfast Area. These outbreaks are small when considered in proportion to the population of approximately 400,000 people in the greater Belfast area. The source of the first outbreak is believed to have been livestock runoff, the second source is believed to have been human sewage from a septic tank, and the third is believed to have been wastewater from a blocked drain (Glaberman et al. 2002, p. 631). Chlorine is commonly used in Ireland, but in this case it is again proven that chlorine is ineffective against the *Cryptosporidium* pathogen. Filtration was in place in the third outbreak, but the blocked drain would have allowed the contaminated water to enter the finished water supply. Ireland primarily relies on surface water sources for drinking water, and these sources are vulnerable to contamination because of frequent heavy rainfalls and the large numbers of farms with livestock (Zintl et al. 2009, p. 271). This combination poses a serious threat to the safety of drinking water. Although high numbers of livestock are a major concern, only the first outbreak was caused by *Cryptosporidium* of an animal genotype, while the second and third were caused by *Cryptosporidium* of a human genotype (Glaberman et al. 2002, p. 632). With the knowledge of a high likelihood of contamination, whether due to animal or human fecal matter, monitoring and treatment in the area would be necessary in order to avoid future outbreaks. The three outbreaks reflect poorly on the ability of Ireland's water system to monitor and treat their water effectively.

In April 2001 an outbreak of cryptosporidiosis occurred in North Battleford, Saskatchewan. This outbreak caused between 5,800 and 7,100 cases of illness, in the city of approximately 15,000 people. The surface water source of North Battleford is the North Saskatchewan River, which at the time had no protection programs established to prevent source contamination. The treatment at the plant included both chlorination and filtration. It is believed that the source of contamination was sewage from a sewage treatment plant 3,500 m upstream from the intake of the drinking water plant (Hrudey et al. 2002, p. 401). The sewage treatment plant was reported as not meeting operating standards due to old equipment and inadequate practices (Woo and Vicente 2003, p. 261). Another possible source of contamination could be calf feces runoff from the agricultural activity in the area in combination with heavy spring rainfall (Woo and Vicente 2003, p. 261). Again, treatment failure was the main problem within the water system that should have been able to prevent the outbreak. Inadequate coagulation, which is part of the filtration process, was the cause of the outbreak. Also a lack of knowledge and education on the topic of water treatment, particularly concerning the specific pathogen *Cryptosporidium*, was determined to be an issue concerning the capabilities of the plant staff (Woo and Vicente 2003, p. 262). Overall the North Battleford outbreak revealed a variety of problems that allowed the *Cryptosporidium* pathogen to enter the drinking water system.

Cryptosporidiosis then occurred in Gwynedd and Anglesey, Wales in November 2005. Lake Cwellyn was the surface water source for the reservoir that supplied water to approximately 70,000 households. There were 231 cases of illness caused by this outbreak. At first, runoff of animal fecal matter was suspected because of heavy rains prior to the outbreak, but this was not unusual weather for this area. The source of contamination is believed to have been human sewage that entered the reservoir from a sewage treatment system. The human strain of *Cryptosporidium* is more dominant in the autumn, while the animal strain is more frequent in the spring. The treatment of the drinking water included pressurized sand filtration and chlorination, but these methods were not designed to be effective against *Cryptosporidium*. There was no specific treatment failure, but action was taken to add more treatment to the system. UV treatment was installed, and when it was operating effectively, the boil water advisory was removed (Outbreak Control Team 2006, p. 7). This is the largest waterborne outbreak of cryptosporidiosis in Wales. The early issuing of a boil water advisory on November 25 probably contained this outbreak. Only a small proportion of the population became infected, and the installation of UV treatment is a strong preventative measure against future occurrences.

Another outbreak of *Cryptosporidium* occurred in Galway Ireland in February 2007. The outbreak caused approximately 242 cases of illness in the city of approximately 72,000 people. Two treatment plants are used to treat the water supplied from Lough Corrib (a lake) to the city of Galway. One treatment plant was newer and used coagulation and rapid gravity filtration, and the second was older and had no filtration. The water of the two plants is mixed and then distributed to the consumers. The source of the contamination is believed to have been human fecal matter, but the source has not been confirmed. Boil water advisories were issued by four water suppliers that use water from Lough Corrib. Treatment failure from a lack of filtration in the second plant is likely to be the cause of the outbreak. Closures and upgrades have occurred since the outbreak.

Outbreaks of cryptosporidiosis have occurred in both small and large communities, as shown through the history of drinking water outbreaks. This indicates the strength and ability of the *Cryptosporidium* pathogen to overcome standard treatment of water systems of small rural communities, but also urban areas, as seen in Kelowna, Kitchener/Waterloo, Milwaukee, and Carrollton. System failures, particularly in the treatment process, are the main contributing factor that allows outbreaks to occur. Ineffective chlorine treatment, and a lack of or inadequate filtration for surface water sources are the common elements in the outbreaks of cryptosporidiosis.

2.2.2 *Giardia*

Giardia is another common cause of drinking water disease outbreaks. Similar to *Cryptosporidium*, *Giardia* is also a protozoan that causes symptoms of diarrhoea and abdominal pains (Craun 1979, p. 819). *Giardia* is resistant to minimum levels of chlorine disinfection, and therefore higher concentrations and longer contact times are required for effective treatment, especially in cold water where resistance increases further (Betancourt and Rose 2004, p. 224). Therefore, a reliance on only chlorine disinfection is ineffective and inadequate against the *Giardia* pathogen. Filtration and alternative methods, as with *Cryptosporidium*, are necessary in order to prevent outbreaks.

A significant early outbreak of *Giardia* contamination occurred in November 1974 in Rome, New York. The outbreak in the surface water source caused 4,800–5,300 cases of illness, in the city of approximately 50,148 people. Rome's water supply was from Fish Creek and it is believed that the source of contamination was untreated human waste. At the time of the contamination only chloramine disinfection was used, with no filtration or sedimentation. Chlorine and ammonia were added to the water entering the reservoir, which forms chloramine, and chlorine was added again to the water leaving the reservoir (Shaw et al. 1977, p. 428). Disinfection as the only treatment method is insufficient in preventing outbreaks of giardiasis (Craun 1979, p. 818).

In September–December 1979, an outbreak of giardiasis occurred in Bradford, Pennsylvania affecting 3,500 people. The treatment system for the surface water source included chlorination, but not filtration. Again, in this case minimum levels of chlorine were ineffective against the *Giardia* pathogen. The source of contamination is believed to have been fecal matter from beavers in the watershed. Beavers are common carriers of *Giardia*. Inadequate treatment and monitoring is believed to have caused the spread of the outbreak (Hrudey et al. 2002, pp. 402, 404). Insufficient levels of chlorine that were unable to provide a chlorine residual in the distribution system, the lack of filtration, and the failure to monitor chlorine residual levels allowed the outbreak to occur. Following the outbreak the municipality built a treatment plant with filtration in an effort to prevent future outbreaks.

Another *Giardia* contamination occurred in December 1985 in the water reservoir in Pittsfield, Massachusetts. The outbreak caused 3,800 cases of illness among the population of 50,265 people (Hrudey et al. 2002, p. 399). The source of contamination is believed to have been fecal matter from infected beavers or muskrats. The cause of the outbreak was due to water treatment changes at the treatment plant. Prior to the time of contamination, the city used two surface reservoirs that were chlorinated, but not filtered (Kent et al. 1988, p. 139). During this time a new filtration system was in the process of being installed on the first reservoir, and so a third reservoir was brought online to phase out the use of the first reservoir while filtration was being installed. There was an increase in turbidity in the third reservoir; to make matters worse, chlorine treatment levels were low during that time because of a malfunctioning chlorinator. Therefore, the water was extremely vulnerable to contamination because of lack of disinfection and the filtration had not yet been installed. Following the outbreak the system was hyper-chlorinated and flushed, and chlorine residual levels and contact times were also increased.

In 1986, Penticton British Columbia experienced a drinking water outbreak of over 3,000 cases of giardiasis. The mixed water source, of both ground and surface water, was chlorinated but unfiltered. The source of the contamination is believed to have been animal fecal matter that entered the water through a spring runoff (Hrudey et al. 2002, p. 400). Treatment failure was the main contributing factor for this outbreak. Filtration, which was not used, is necessary against *Giardia* because of its resistance to minimum chlorine levels.

Treatment failure is the common thread among the outbreaks of giardiasis, just as with cryptosporidiosis. In all the included outbreaks of giardiasis, a lack of filtration is a common factor that enabled the pathogen to spread through the distribution system to the consumer. Outbreaks of giardiasis occur most commonly in communities that rely on surface water. Again, as with cryptosporidiosis, the outbreaks of giardiasis also occurred in both small and large communities. Therefore, these outbreaks show the need to provide effective alternative treatments, instead of relying on chlorination alone.

2.2.3 *Toxoplasma*

Toxoplasma gondii is a rare microbial pathogen that has only caused three recorded drinking water outbreaks. The first occurred in Panama in 1979, the second in Victoria, British Columbia, Canada, in 1995, and the third and largest outbreak occurred in Brazil in 2002 with 209 cases of illness (Dumetre and Darde 2003, p. 654). *Toxoplasma* is resistant to the usual methods of chlorine treatment, but because it is rare, water systems have not been as alert as they should have been.

The outbreak in Victoria, British Columbia occurred from October 1994 to April 1995 in Humpback reservoir. Victoria is supplied by the Humpback reservoir and the Sooke reservoir. The outbreak caused 110 reported cases of illness, although it is believed that the contamination infected 2,900–7,700 people (Aramini et al. 1999, p. 306). The source of the contamination is believed to have been cat or cougar fecal matter (Aramini et al. 1999, p. 307). *Toxoplasma gondii* is also a protozoan pathogen, and therefore is resistant to chemical disinfection, which allows the contamination to spread throughout the water system and into the taps of consumers. The water system in B.C. relied on chloramine disinfection without filtration, enabling the survival of the pathogen within the reservoir (Dumetre and Darde 2003, p. 654). This was Canada's first and only reported outbreak of toxoplasmosis, and is also the first outbreak of toxoplasmosis in a developed country.

2.3 Bacteria

2.3.1 *Campylobacter*

Campylobacter jejuni is another cause of drinking water disease outbreaks. *Campylobacter* also causes gastroenteritis illness similar to *Cryptosporidium* and *Giardia*. The main symptom found in humans infected with the *Campylobacter* pathogen is diarrheal illness. However, *Campylobacter* is a bacterial pathogen, unlike *Cryptosporidium* and *Giardia* that are protozoan pathogens. Chlorine disinfection is effective against bacteria, and therefore outbreaks of *Campylobacter* should be easily preventable. In theory, there should be very few outbreaks since chlorination is the most common chemical disinfection. However, system failures in treatment processes are the main contributing factors that allow outbreaks to occur. *Campylobacteriosis* is more commonly a food borne disease found in raw or undercooked poultry, but there have been several waterborne outbreaks of significance.

In May 1983 in Greenville, Florida, an outbreak of *Campylobacteriosis* occurred. The ground water source supplied the rural community of 1,096 people. Animal fecal matter was determined to be the source of the contamination, causing 865 cases of illness (Sacks et al. 1986, p. 424). The *Campylobacter* pathogen entered the water source through infected bird droppings into open water towers. The system was reported to have other deficiencies, in addition to the open towers, which allowed this contamination to spread undetected. These included an unlicensed operator and insufficient treatment (Sacks et al. 1986, p. 424). The treatment of the system included pre-chlorination, flocculation, and post-chlorination. With effective chlorination the outbreak should have been prevented, but the levels of chlorine in this case were insufficient. The pre-chlorinator failed and the water backed up into the post-chlorinator, which was not effectively chlorinating the water before it entered the underground well. Equipment, operational, and treatment failures all contributed to this outbreak. If the plant had been properly maintained, the outbreak would have been prevented.

In March 1985 the groundwater source of Orangeville, Ontario became contaminated with *Campylobacter*. The outbreak caused 241 cases of illness. The source of the contamination was surface drainage from farming activity that followed a heavy spring rainfall and runoff (Hrudey et al. 2002, p. 399). The treatment of the system did not include chlorination because it was not required at the time for the deep wells. A lack of treatment, especially when considering the proximity of nearby animal farming, in combination with heavy rainfall and runoff, allowed the spread of the outbreak (Hrudey et al. 2002, p. 399). Treatment failure is again shown as the cause of a drinking water outbreak. Following the outbreak, chlorination disinfection has been installed.

In 1998, a groundwater source became contaminated with *campylobacter* in the Haukipudas municipality in Finland. The area of 15,000 people suffered approximately 3,000 cases of illness. The source of the outbreak is believed to have been bird droppings through holes in the water tower. The water supply was not chlorinated; treatment failure from a lack of chlorination is again the major contributor that caused this outbreak to occur.

Outbreaks of *Campylobacteriosis* are not as common as cryptosporidiosis or giardiasis, but *Campylobacter* is still considered a threat to the safety of drinking water. Proper chlorination or other form of disinfection would be effective against the campylobacter pathogen.

2.3.2 Escherichia Coli

Escherichia coli is a well-known drinking water contaminant because of the highly publicized drinking water contamination that occurred in Walkerton, Ontario, Canada in 2000. *E. coli*, similar to *Campylobacter*, is a bacterial pathogen that is not resistant to chlorine disinfection. *E. coli* contamination could be easily prevented particularly in smaller rural communities, since chlorine is the most commonly used chemical disinfection for drinking water. Groundwater in the Walkerton case became contaminated with several pathogens, but primarily *E. coli* and *Campylobacter jejuni*, causing over 2,300 cases of illness and seven deaths in the community of approximately 4,800 people (Hrudey et al. 2002, p. 397–398). The source of the contamination was cattle manure. It occurred through a combination of heavy rainfall causing the runoff into the water source, and also system deficiencies such as human error; the water system managers did not detect the contamination, and therefore did not treat it. The Walkerton Inquiry was commissioned following the outbreak, and was released in 2002. The Inquiry discussed the reasons and causes of the outbreak, and also provided recommendations for new and existing legislation to prevent future occurrences. The Inquiry emphasized the necessity of the multi-barrier approach to provide safe drinking water. A common theme among the research on drinking water outbreaks is the failure of barriers in water systems, allowing contaminations to occur and pass through distribution systems to cause illness. System deficiencies, including treatment and operational failures, were the main reasons for the Walkerton outbreak of 2000.

Another case of *E. coli* contamination occurred earlier in Cabool, Missouri in 1989, prior to the Walkerton contamination. The small rural community of approximately 2,090 people reported 243 cases of illness and four deaths. Although not as publicized as the Walkerton incident, Cabool's outbreak was also significant as it also caused death. Cabool uses a groundwater source, but direct source contamination was not believed to have occurred. The source of contamination is believed to have been fecal contamination from sewage. It occurred from a lack of disinfection following replacement of water meters and repairs to broken water mains. Unseasonably cold weather caused the water mains to break. The sanitary sewer system was also vulnerable to storm runoff (Hrudey et al. 2002, p. 403). The introduction of chlorination into the system subdued the outbreak (Rice et al. 1992, p. 38). Again, similar to Walkerton, this outbreak was due to system deficiencies. Treatment, particularly disinfection, is critical to ensure safe drinking water. Proper chlorination would have been effective against the *E. coli* contamination and would have prevented the outbreak.

In both outbreaks of *E. coli*, loss of life was associated with the contamination. The severity of the outbreaks emphasizes the need for adequate treatment, operational practices, and the maintenance and upkeep of equipment. A common thread between the bacteria pathogens, *Campylobacter* and *E. coli*, is that outbreaks most commonly occur in communities with surface sources or groundwater supplies that can come under the influence of surface water and there is inadequate disinfection.

2.4 Lessons from Disease Outbreaks

Drinking water disease outbreaks are the result of multiple failures within a water system. The most common failures that allow outbreaks to occur are improper or neglected treatment and failure to monitor operations. Outbreaks indicate the need for continual vigilance and adequate monitoring in the drinking water production and distribution, as well as continual testing of water quality to maintain adequate quality standards. Outbreaks can be used to gain knowledge and understanding of the techniques and methods that are most effective for providing safe drinking water. Lessons can be learned nationally within countries as well as internationally among countries, as shown here from Canada, the United States, and Europe.

Steven and Elizabeth Hrudey are able to make conclusions in their book, *Safe Drinking Water* (2004), based on their summary of outbreaks from 1974 to 2002. They conclude that the multi-barrier approach continues to be a requirement for a safe drinking water system. Barriers in place at each stage within the system for the source, treatment, distribution, monitoring, and response are all required to ensure safe drinking water. Both human and nonhuman elements can cause failures throughout the system. Continued emphasis on the multi-barrier approach is necessary in order to detect and treat contamination at all stages before the water is distributed to the consumer. This approach is still the most effective method to provide safe drinking water.

The barrier of treatment is critical to the overall process. If an unknown contamination occurs, the goal of treatment is to inactivate and/or remove the pathogen before the water continues into the distribution system. Chlorine is the most commonly used chemical disinfectant because of its cost-effectiveness. We know that standard chlorine disinfection is effective against bacterial contaminations of *Campylobacter* and *E. coli*, but ineffective against protozoan contaminations of *Cryptosporidium*, *Giardia*, and *Toxoplasma*. *Cryptosporidium* is the most resistant, but all three protozoa are able to surpass simple chlorine treatment, which has been shown to cause numerous outbreaks. In the wilderness it may not be possible to prevent contaminants from entering water sources, especially surface water, but the barriers of filtration and disinfection are critical in preventing the spread of contamination that lead to outbreaks.

Hrudey and Hrudey also conclude that microbial pathogens are the primary concern for drinking water safety. All the included outbreaks are caused by pathogens, thus indicating the longevity and persistence of the problem and their dominance among contaminations. Pathogens threaten the safety of drinking water because of the possibility of contamination anywhere throughout a water system and their ability to surpass the treatment process. Pathogens originate from within human and animal fecal matter. Sources deemed to be of high quality could become contaminated with such matter, especially surface water sources. Hrudey and Hrudey emphasize the growing occurrence of *Cryptosporidium* since the 1990s up to the Walkerton contamination in 2000. With its high resistance to chlorine, the most commonly used method of treatment disinfection, the threat of *Cryptosporidium* continues past the Walkerton outbreak to pose the highest risk to water systems. With the extent of research and the numerous outbreaks associated with this dominant pathogen, it is surprising that outbreaks continue to occur.

The Hrudeys also emphasize the effects of a *change* on a drinking water system. A system that is adaptable to change will be more capable of providing safe drinking water. Change can include changes in the weather, changes within the community, and changes within the water system. This is a contributing factor in many of the mentioned outbreaks. Change in the weather, either due to season changes or severe rainfalls associated with climate change, prior to the occurrence of outbreaks is a common event, such as in the outbreak of Carrollton Georgia in 1987, Warrington England in 1992, Cranbrook B.C. in 1996, and Galway Ireland in 2007. Change in a community can occur from human activity, such as farming. Agricultural runoff from farming activity was the specified cause in outbreaks such as Jackson County Oregon in 1992, Warrington England in 1992, and in Galway Ireland in 2007. Change in a water system contributed to outbreaks such as Kitchener in 1993 when the water system switched from a groundwater source to a surface source, and also in Pittsfield Massachusetts in 1985 when a filtration plant was in the process of being installed. Change should act as a warning to system operators of possible contamination. Monitoring should be heightened during times of change, and precautions may be necessary.

The conclusions by Hrudey and Hrudey (2004), based on outbreaks prior to 2002, emphasize that the Walkerton outbreak should have served as a major landmark in the history of contaminations. However, it does not seem that water authorities have absorbed lessons from that outbreak, as outbreaks have continued to occur since then. The conclusions and lessons described by the Hrudeys in their book can therefore be further expanded with new information by including outbreaks after 2002. The outbreak in Gwynedd and Anglesey, Wales in 2005 and the outbreak in Galway Ireland in 2007 are the most recent outbreaks. Including these cases provides the opportunity to consider whether outbreaks have changed patterns after Walkerton. Considering the patterns among the outbreaks is important in determining what factors contribute to the occurrence of an outbreak.

From the analysis of outbreaks reported here, we can conclude that there are no seasonal patterns to outbreaks. Contaminations have continued to occur during spring runoff from winter thaws and with higher amounts of rainfall. However, outbreaks can occur at any time during the year. This can be observed from the outbreaks reported here, as over half of the outbreaks surveyed here did not occur in the spring season. Spring runoff and rainfall are natural events, but improper practices by system operators can also cause outbreaks. Frequent human failures that cause outbreaks include improper and ineffective treatment, insufficient monitoring, and inadequate training of operators.

We can also conclude that outbreaks do not follow a pattern based on the size of a water system. In contrast to what may be a common belief, outbreaks are not specific to only small rural areas. Although outbreaks may be more frequent in smaller towns because of lower maintenance and less efficient water systems, due to lack of finance, this does not mean that larger systems are immune from failure.

This survey of waterborne disease outbreaks makes it clear that outbreaks can occur in both large and small communities. As mentioned above, Milwaukee with a larger population of 840,000 served by the water system and North Battleford with a smaller population of 15,000, have both experienced outbreaks of cryptosporidiosis. The major difference is that an outbreak among a larger population is likely to have a more significant impact, as more people are affected. The outbreaks that have occurred after Walkerton in 2000 also indicate that large communities are also susceptible to contamination.

Another factor that seems to stand out in the outbreaks is the reliance on chlorine. Communities that rely heavily and especially those that rely solely on chlorine disinfection are vulnerable to contamination. This chapter suggests that the most dominant microbial contaminant of the outbreaks referred to here is a protozoan pathogen, namely *Cryptosporidium*. The ability of protozoa to infiltrate and pass through many drinking water systems is because of their resistance to chlorine. Alternative treatments that are effective against protozoa are filtration, ozone treatment, and UV light treatment. Communities that have experienced problems of *Cryptosporidium* contamination often rely heavily or solely on chlorination. This is a significant limitation in the use of chlorine. Another disadvantage of chlorine is the byproducts that can result from its use. Trihalomethanes (THMs) form through a reaction between chlorine and organic compounds. These are known as disinfection by products (DBPs) and can have long-term health effects (Moghadam and Dore 2012). Chlorine can also create a distinctive taste if high levels of disinfection are required, which is often strongly disliked by receiving communities. Alternative methods of disinfection should be considered to avoid the problem of ineffective disinfection and the occurrence of THMs.

Multi-use watersheds involve a variety of activities and operations that could all contribute to a contamination. It is clear that farming operations often result in animal fecal matter contaminating water courses. Sewage treatment plants are another common cause of contamination. Animal and human fecal matter is the most common source of contamination in the drinking water outbreaks. The increase of human activity in a watershed also increases the possibility of contamination. Consequently multi-use watersheds need to increase the scrutiny of their water quality.

Boil water advisories (BWAs) are issued in order to prevent disease and drinking water outbreaks. A BWA requires all citizens of the specified community to boil their water prior to consumption in order to kill the possible pathogens within the water. BWAs are issued at the local level by the water authority of a community often following the detection of contamination, as a precautionary measure. Used wisely a BWA can prevent an outbreak. A BWA is effective when the detection of a pathogen in a water system is confirmed. However, a BWA can be ineffective when uncertainty of the water quality results in a continuous use of BWAs. A BWA is also ineffective when it is used as an alternative by water systems rather than providing the necessary treatment and equipment maintenance to be able to supply safe drinking water. A BWA should not be issued to avoid the responsibility of proper treatment and maintenance, but continuous and longstanding advisories indicate that this occurs, particularly in small communities. Continuous BWAs are more common in smaller rural communities that are unable to maintain or upgrade their systems, often due to financial inability. Many residents are known to disregard a BWA, as some BWAs last many years.

Overall, the issue of drinking water quality will continue to remain a primary concern worldwide. Contamination and outbreaks can occur at any time and anywhere regardless of season or size of water system. The patterns among the outbreaks clearly show the ineffectiveness of chlorine against the threatening *Cryptosporidium* pathogen, the vulnerability of multi-use watersheds, and the failures of BWAs that are often overused. There is no substitute to proper treatment for safe drinking water.

What is the first step in preventing waterborne disease outbreaks? In the multi-barrier approach, the first component is the establishment of protection of source waters. This may require a watershed protection plan, including legislation to support watershed protection in the law of the land. Implementing watershed protection requires an understanding of the key principles of watershed management. The next section provides a succinct statement of the principles; the risk management component of watershed management is covered in Chap. 6.

2.5 Principles of Watershed Management

A watershed is an area within which all water bodies such as rivers and streams accumulate and eventually find an outlet. A watershed may also have one or more sub-watersheds. Watersheds are naturally cohesive hydrological units, encompassing a large area of land. The successful management not only aids the hydrological system, but also benefits the socio-ecological entity. We now summarize the core principles based on successful watershed management in the past.

1. A Good Understanding of Natural Ecosystems

Watersheds are defined by the topographic boundaries including natural ecosystems and urbanized landscapes, or elements of both (United States Environmental Protection Agency (USEPA) 2013, p. 7). The natural processes refer to the dynamic physical and chemical interactions, which form the landscape of the watershed, as well as its water quality According to their characteristics, watersheds can be classified into three management zones (USEPA 2013, p. 8):

- *Upland zones* are land areas above high water level that intercept and transport rain or storm as groundwater.
- *Water-body zones* are surface water bodies, such as stream, river, and ocean, which provide the living environment for aquatic and terrestrial birds and mammals.
- *Riparian zones* are border surface water bodies that filter the surface water runoff.

The communities of humans, plants, and animals rely heavily on the watersheds, but to some extent damage them at the same time. It is an interaction effect between human activities and natural forces, which directly or indirectly changes the conditions of the water and land. More especially, with urban impervious surfaces such as roads and highways, storm-water flowing across the surfaces picks up contaminants that are carried directly in the storm-water drains and eventually enter the watershed (USEPA 2013, p. 13). In addition, the discharged and untreated water carries pollutants such as fertilizers, motor oil, PCBs and heavy metals, which also end up in the watershed; it impacts the water quality as well as public health. Thus, a good understanding of natural systems helps to achieve a harmonious relationship between human activities in the watershed and natural processes.

2. A Watershed Management Framework with the Involvement of Partners and Stakeholders

Building a watershed management framework is necessary in order to prevent environmental problems in advance. The framework describes the goals or problems and outlines the protective actions. Essentially, the framework focuses on a continued process for partners and stakeholders to work together and supports the watershed plans (USEPA 2013, p. 17). These partners and stakeholders make decisions on all the aspects of the framework that includes (a) resource standards (i.e. water quality standards), (b) watershed management approaches, and (c) watershed management projects. Eventually, the coordinated efforts in watershed management facilitate the development of the environment and the economy. For example, in 1992, three major chemical companies (Amoco, Dupont, and Bayer) collaborated with US fish authorities and other professional associations and even local citizens, in reconstructing successfully the ecosystem in the Cooper River region, and also enhanced local economic growth (USEPA 2013, p. 5). Moreover, using sound science in watershed management helps to achieve sustainable goals. For example, in trying to relieve the pressure of water demand due to increasing urban population, adequate water is required for the future in a sustainable manner. Furthermore, making use of scientific management approaches such as sustainability analysis and other tools to improve water productivity will be required to satisfy the water demands in the future.

3. Continuous Improvement Based on the Integrated Watershed Management

The overlapping of multiple jurisdictional boundaries in a watershed and the various environmental and economic interests of stakeholders result in a complexity of watershed management, and thus an integrated management approach applied to the watershed is required to improve the effectiveness of management (USEPA 2007, p. 1). To meet multiple objectives, integrated management refers to all stakeholders utilizing their respective disciplinary approaches to address the priority problems within a given watershed. Specifically, a government agency may be responsible for implementing a watershed management plan, as well as assessing and managing water quality and supply, while a local watershed association may be interested in solving a sedimentation problem in a small watershed, or making sure farming in the region does not have an adverse impact on water quality in the region. A good integrated management approach should connect all the initiatives and actions of government agencies as well as local watershed associations (USEPA 2013, p. 17). Additionally, the process of integrated management is continuous, cyclical, and endless; it includes data gathering, assessment, targeting, implementation, and monitoring (USEPA 2013, p. 15).

4. Flexible Approach in Watershed Management

The watershed management approach is not one size fits all. Because each watershed landscape is shaped by a blend of climate, geology, hydrology, soils, and vegetation, a targeted approach should be applied to support the watershed management depending on different regions of the country (USEPA 2013, p. 20). In practice, watersheds can change over time, due to (for example) the emergence of serious diseases or a change in water flow patterns or due to a change in use patterns. The objectives and approaches of watershed management should be adjusted to adapt to changes in water and land use.

5. Application of Ecological Risk Assessment to Watershed Management

The US has strong regulations on point source pollution such as farms with animals. With the reduced impact of point source pollution (e.g. cattle manure from a specific farm) on source water quality via multiple legislation or regulations in the US, the issue of controlling nonpoint source pollution (pollution sources that cannot be identified) is becoming increasingly important both environmentally and economically (USEPA 2007, p. 6). Due to the limitations of being able to find pollution sources and pathways, nonpoint source pollution problems are not being corrected by existing regulations (USEPA 2007, p. 6). For this reason the EPA recommends an alternative approach, which is the application of Ecological Risk Assessment (ERA) to watershed management. In practice, the primary principles of watershed ERA assist the watershed managers to make decisions on such factors as total maximum daily load of contaminants, resource planning, and land use zoning, and how to mitigate these expected harmful effects. Since the sources of pollution are not known, it is a matter of assessing the risks that emerge from a variety of causes. In the mid-1990s, USEPA's Risk Assessment Forum and the Office of Water collaborated on testing the application of the ERA approach and recorded all the details during the experiment by choosing five watersheds that possessed abundant ecological resources, available dataset, and multiple stressors (USEPA 2007, p. 6). According to the EPA Risk Assessment Forum and the Office of Water, several researchers found that the application of ERA had been beneficial for watershed management (USEPA 2007, p. 6). A complete description of ERA is given in Chap. 6, which covers all aspects of risk management.

The key issue in the risk assessment of point source pollution is an adequate record of agricultural and commercial activity that could impact water courses within the watershed. Where the agricultural activity involves a large number of animals (for example cattle or pigs), there have to be clear animal manure management plans that ensure that none of the manure ends up in the water courses. For nonpoint sources (for example wild animals or birds dying in water bodies at unknown locations, or contaminating the water with their feces), a good overview of the wildlife is necessary, and risks of contamination must be estimated. It would also help if there were water quality monitoring stations, so that when a quality problem arises, the authorities can try to identify the location of the contamination and remove it. If the contamination shows up in groundwater, it may be next to impossible to trace the source of the contamination. But if the frequency has been calculated, then risk assessment methods can be used and "high risk" areas can be mapped to warn local residents of the dangers of contamination of unknown origin.

In the US, a set of best management practices for animal farming are mandatory for large farms. For nonpoint source pollution, the US uses the method of Ecological Risk Assessment, on which more details are considered in Chap. 6. The principles of watershed protection outlined above should be reflected in legislation, wherever these principles are taken seriously. In other words, "voluntary" watershed protection is impossible and all watershed protection practices should be embodied in legislation.

2.6 Conclusion

The key lessons are: (a) a water system cannot rely exclusively on chlorine; (b) water systems must institute careful monitoring of conventional water treatment technology, particularly of flocculation and of chlorination; (c) water systems must be vigilant over the possibility of animal or human fecal material seeping into the water supply at all stages; (d) there is a need to institutionalize a regular protocol of sampling of water quality and the reliability of such sampling, and (e) a determined policy of continual modernization of all components of the complete water treatment train, by investing in newer and safer treatment technologies (see Chaps. 3 and 4).

As a first step in preventing waterborne disease outbreaks, institute a multi-barrier approach, of which the first component is the establishment of a sound watershed management plan that prevents contamination of water courses. The second step in the multi-barrier approach is a clear understanding of drinking water treatment technologies, which is the subject of Chaps. 3 and 4.

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Part II
Drinking Water Treatment Technology and Pricing

In Part II, we attempt to answer the following questions:

- As reliance on chlorine is inadequate, what are the other treatment technologies and their respective average costs, for any given scale of water production?
- Can we classify these treatment technologies on the basis of what contaminants they can remove?
- Does the removal of more contaminants always lead to higher average costs, for a given scale of water production?
- Is there a treatment technology that is the "state-of-the-art," in that it can remove almost all contaminants at an affordable cost?
- What are the theoretical mechanisms of pricing water?
- What are the main actual pricing mechanisms used in the developed countries?
- Can fairness, equity, and other moral judgments be embedded in the way water is priced, for different populations, such as the rich and the poor?

Chapter 3
Water Treatment Technologies and Their Costs

3.1 Introduction

According to the USEPA (1996), 94 percent of 156,000 public water systems in the US are small water systems, serving a population of fewer than 3,300 people. In Canada, the proportion of small systems in one survey was over 75 percent (Environment Canada 2004). With a smaller tax base all small water systems face special challenges, unless the government aggressively supports small water treatment systems. In Canada, many continue to encounter boil water advisories and even disease outbreaks. No doubt that with appropriate public funding, many of these problems can be reduced or eliminated. However, typically in North America, each small community or rural jurisdiction must cover its own capital and operating costs of their drinking water supply, although some jurisdictions offer a subsidy for capital costs. Often a rural community has a small population, lower average income, and consequently a lower tax base. These financial constraints as well as other risk factors were highlighted at a 2004 Montana conference on small water systems (Ford et al. 2004). These constraints are even more severe in developing countries.

Threats to public health persist in rural and small water systems even in the most advanced high-income countries like the USA, Canada, and Europe. The factors accounting for some of these waterborne disease outbreaks were explored in Chap. 2. The objective of this chapter is to present statistical models of costs based on new data obtained from manufacturers of a menu of treatment technologies suitable for water systems, small and large; this information would be of use to local government officials, water engineers, and planners. Most of these technologies are concerned mainly with plants that rely on surface water as the source.

For the USA, the American Water and Wastewater Association (AWWA) has published a number of reports that include recent water utility survey data on current disinfection practices and operations compared with practices in the late 1970s (AWWA Water Quality Division Disinfection 1992 and 2008). According to the AWWA 2008 report, chlorine gas remained the predominant disinfectant, used

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by 63 percent of respondents, whereas those who used chloramine accounted for 30 percent; chlorine dioxide for 8 percent; ozone for 9 percent; and ultraviolet light (UV) for 2 percent. The comparable figures for Canada are also available: according to the Environment Canada survey of Municipal Water and Wastewater Plants (2004), in Canada there were 2,402 drinking water systems in that survey, of which 1,513 reported a population of fewer than 3,000. Of these 1,513 drinking water plants, 136 gave information on the type of disinfection technology they use. Some 93 per cent (127 out of 136) used chlorine as the only disinfectant. Those using UV or ozonation accounted for only 6 percent of the total. There is a potential for improving water quality by adopting newer technologies such as UV or ozonation and reducing the probability of waterborne disease outbreaks. At the same time, there is an enormous market potential for corporations that can sell a competitive technology that is also cost-effective. The rest of the chapter presents average "first approximation" cost per cubic meter based on statistical modeling on recently collected data on costs for different flow rates from equipment manufacturers in North America. We show that there exists a menu of cost-effective technologies that might be considered for possible modernization of water treatment plants, including small water treatment plants.

We classify these technologies into six classes, depending on the contaminants removed. Our statistical results show that average costs (including capital, operating, and maintenance) of production of these technologies depend on the flow rate as well as the number of contaminants removed. The larger the flow rate is, the lower the cost will be per unit of volume treated, and the more the contaminants removed are, the higher will be the cost, for any given flow rate. One of our major findings is that for surface waters except those with high color and turbidity, UV-based treatment technologies are cost-effective. However, for any particular system, water engineers would take site-specific features into account to determine what technology is most appropriate.

This chapter is organized as follows. Section 3.2 of this chapter presents a scheme, which classifies water treatment technologies based on the contaminants they remove; Sect. 3.3 shows projected costs of four technologies, which are Ultra Violet disinfection (UV), Micro filtration—ultra filtration (MF-UF), High rate Clarification & Filtration (HRC), and Ozonation; Sect. 3.4 is an analysis of the costs of Advanced Oxidation Processes; Sect. 3.5 makes brief reference to Reverse-Osmosis (RO) and Nano-Filtration (NF), but there is a whole chapter devoted to RO and other more advanced treatment technologies (Chap. 4). In Sect. 3.6, we present examples of costs of actual existing small water treatment systems in Canada. Finally, Sect. 3.7 is a general summary with concluding remarks. Our major conclusion is that for surface water sources except those sources with high color and/or high turbidity, UV is a competitive and viable treatment technology that should be considered in a menu of suitable technologies. However, as stated before, the actual adoption of a treatment technology depends on many site-specific features (such as location, distance from major cities, and topography) that are best determined by the consulting engineers.

3.2 Six Classes of Water Treatment Technologies

Suppose we consider a large state-of-the-art water treatment plant and use their costs of water treatment as an initial benchmark. One such treatment plant is the Seymour-Capilano Filtration Plant run by the Greater Vancouver Regional District (GVRD) in Canada. This plant, which came on stream in December 2009, will give us a perspective on costs at a large water plant. The source water for this GVRD plant is of high quality, and is possibly free of micro-pollutants, largely because of the source water quality. Table 3.1 gives some information on this system. Due to economies of scale, the plant has the potential to produce drinking water at CAN $0.40 per cubic meter. However, when the distribution costs are added, it is estimated that the consumer would pay about $1 per cubic meter. This provides a comparative benchmark of the costs at a large state-of-the-art water treatment system and shows to what extent the costs of small water systems differ from those at a large system.

Not all systems can produce at the cost and level of drinking water quality that this Vancouver plant can produce. But our survey of new technologies suggests that there are technologies for small systems with similar low average costs per cubic meter. As stated before, in general, costs depend on the *number* of contaminants removed, although there may also be other nonlinearities. Below we provide a scheme, which would allow us to classify a given water treatment plant by the number of contaminants removed, based on technology being utilized at the plant. We postulate six classes of water treatment technologies in Table 3.2.

Class 1 represents the minimum level of treatment, which is disinfection by chlorination only. We consider chlorination the minimum disinfection treatment level since all water treatment plants are required to produce water that is free of pathogens. While most groundwater based systems would rely on chlorine only (Class 1), many surface water small water systems will be Class 2, i.e. water that has suspended solids removed and is disinfected. In a Class 3 plant, protozoa will also be removed or inactivated, possibly with the aid of UV or ozonation. If, in addition, all dissolved organic matter is also removed before chlorination, then that would be water without disinfection byproducts (DBP), and we classify such treatment technology as Class 4.

On the other hand Class 5 (i.e. Classes 5a and 5b) represents technologies that also remove chemicals, micro-pollutants, DBPs, protozoa, and suspended solids, in addition to disinfection. In the scheme proposed in Table 3.2, each progressively

Table 3.1 Description of GVRD state-of-the-art water treatment plant

Parameter	Description
Capital cost	$1 billion
Capacity	1,900,000 m^3/day
Break-even cost	$0.40/$m^3$
Treatment system	Sand Filtration, UV and hypochlorite

The information in Table 3.1 is from personal communication

Table 3.2 Proposed water treatment classes

Class	Typical treatment technology	Contaminants removed
Class 1	Chlorination	Water disinfection; removal of most pathogens
Class 2	High Rate Clarification & Filtration	Disinfection plus suspended solid removal
Class 3	Ultra Violet	Class 2 plus removal of Protozoa
Class 4	Ozonation	Class 3 plus removal of dissolved organic matter (no DPB[a] precursors)
Class 5a	Activated carbon, powdered or granular	Class 3 plus removal of geosmin and other taste and odor compounds, DBPs, Volatile Organic Compounds, Endocrine Disruptors, micro-pollutants, pesticides, pharmaceuticals and personal care products
Class 5b	Advanced oxidation process	Class 5a plus higher efficacy of the removal of chemicals and other micro-pollutants (e.g. pesticides, pharmaceuticals, taste and odor concerns)
Class 6	Reverse Osmosis OR Distillation	Class 5 plus removal of salinity

[a] DPB stands for "disinfection byproducts." See footnote 2

higher treatment class indicates a greater removal of contaminants. However, this classification scheme is fairly broad in scope, an initial attempt, although other more finely graded classifications are possible. Note that we are classifying *treatment categories or classes, not final water quality*. What emerges from this classification is a way of comparing final water quality *indirectly*, on the basis of what treatment systems are used, and also assessing any possible long-term health threats.

In North America, most drinking water comes from surface water, which needs to be treated adequately. The data presented in the introduction to this chapter shows the dominant role played by chlorine and chlorine derivatives in North America, where this Class 1 technology is concerned almost exclusively with the removal of pathogens, although we know that chlorine is not effective against protozoa and other pathogens. However, for most large cities and populations, the conventional water treatment method is coagulation, flocculation, clarification, and filtration, and is typically followed by disinfection by chlorine or chlorine derivative. But the failure of a flocculator led to an outbreak of *cryptosporidiosis* in Carrollton Georgia in 1987; the failure of a chlorinator led to an outbreak of *giardiasis* in Bradford Pennsylvania in 1979. Thus, the conventional treatment train is best described as being Class 3 *if it removes all protozoa*; it cannot be classified as Class 4 as chlorination will leave DBP precursors in the water. For this reason, in Ontario and indeed in the whole of North America, the main DBPs, called Trihalomethanes (THMs), nitrosamines and Haloacetic Acids (HAAs) are regulated with maximum contamination limits. But there are also many other DBPs, called Halides, that are not regulated at all.

The most significant drinking water outbreak of *cryptosporidiosis* was in Milwaukee Wisconsin from March to April of 1993, the worst waterborne disease outbreak in the US history. Two water treatment plants supplying water to Milwaukee used water from Lake Michigan. Both plants used conventional treatment of coagulation, flocculation, sedimentation, rapid sand filtration , and chlorination treatment (Solo-Gabriele and Neumeister 1996, p.81). Again the failure to remove a protozoon indicates that these plants functioned as no more than Class 2 treatment systems.

Based on the evidence and the above classification system, we are led to the conclusion that the conventional treatment plants in North America are at best Class 3, and no more than Class 2 when they fail to remove protozoa. Note that this conclusion is based on treatment technologies and not on the quality of final drinking water, which may be quite good in some areas, depending on the characteristics of the source water; our focus here is on treatment.

It should also be noted that after a large fall in unit costs of ozonation, many water utilities are choosing ozonation[1] as the primary treatment option (Class 4). In Europe the treatment of choice is granular activated carbon, which we classify as Class 5a. Granular activated carbon (GAC) has been used extensively for the removal of dissolved organics from drinking water. In the early 1970s, it was reported that bacteria, which proliferate in GAC filters may be responsible for a fraction of the net removal of organics in the filter. Following this discovery, pre-ozonation was found to enhance significantly the biological activity on GAC. The combination of ozonation and GAC is commonly referred to as the biological activated carbon (BAC) process, or biologically enhanced activated carbon process. This was implemented in many large water treatment plants in Europe in the 1980s (Dussert and Stone 2000). The efficacy of activated carbon in removing all sorts of contaminants has been further confirmed by Rodriguez-Mozaz (2004).

Advanced oxidation processes (with ozonation or UV-based) are essentially the same as Class 5a, but experiments show a greater efficacy of removal of the same contaminants as those in Class 5a; we, therefore, classify Advanced Oxidation processes as Class 5b.

We should also note that for 90 percent of the residents of Ontario, the source water is the Great Lakes, which also receive wastewater that is not always treated to remove chemicals, particularly pesticides, pharmaceuticals and personal care products; this topic is deferred to the chapter dealing with wastewater and its impacts on drinking water.

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[1] Ozone (O_3) and its primary reactive products, the hydroxyl free radical (OH*), are strong oxidizing agents. However, ozonation can also lead to the formation of potentially harmful byproducts that include bromate ions (BrO_3), aldehydes and peroxides. The use of O_3 as an alternative disinfectant to chlorine will not produce chlorinated trihalomethanes (THMs), halo-acetic acids (HAAs) or other chlorinated byproducts; but it can react with natural organic matter (NOM) to produce a variety of oxidation byproducts that typically include aldehydes, aldo- and keto-acids, carboxylic acids and peroxides. However, there are technologies that can be used to minimize these byproducts. For more information on the chemistry of ozonation and how to minimize these byproducts, see: http://www.wwdmag.com/microfiltration/strategies-minimizing-ozonation-products-drinking-water.

In Germany, roughly 74 percent of drinking water is drawn from ground and spring water, and the remainder is drawn from surface water sources, such as lakes and rivers (Althoff 2007). By 2010, 63 percent of the groundwater bodies in Germany had achieved a rating of "good chemical status" (BMU 2014). Of the total 1,000 groundwater bodies, only 4 percent have not achieved a "good quantitative status," i.e. 4 percent of the aquifers did not have enough water. The status of surface water is such that 88 percent of water bodies achieved a "good" chemical status, while only 10 percent of all surface water bodies had obtained at least a "good" ecological status (BMU 2014). Given the quality of groundwater, practically no disinfection is needed. The 2011 Profile of the German Water Sector states:

> The quality of drinking water is so good that the use of disinfectants in water treatment can even be forgone in many places without [compromising] the high hygienic drinking water standard.

Since there is no chlorine, there are no DBPs; in areas where the source is groundwater, there are no chemical residues in the water and of course no salinity. Thus, for the groundwater sources we can conclude that German drinking water from the water treatment plants is equivalent to Class 5. In North Rhine-Westphalia, in the City of Cologne, they use groundwater as the source, which is then filtered through activated carbon, producing a very high quality of water. To quote from the City of Cologne website (RheinEnergie 2013):

> Some waterworks in Cologne used disinfectant to prevent an increase in the number of germs, and thus hygienic deterioration of the drinking water quality on the way to the customer. Our water lab proved, however, that the perfect hygienic quality of drinking water can be guaranteed even without the use of chlorine dioxide or chlorine.

Where surface water is used in North Rhine-Westphalia, they detected perfluorooctanoate (PFOA) in drinking water at concentrations up to 0.64 μ/L in Arnsberg, Sauerland, Germany. In response, the German Drinking Water Commission (TWK) assessed perfluorinated compounds (PFCs) in drinking water and in June 2006 became the first in the world to set a health-based guideline value for safe lifelong exposure at 0.3 μ/L (sum of PFOA and perfluorooctanesulfonate, PFOS). PFOA and PFOS can be effectively removed from drinking water by percolation over granular activated carbon.

For each treatment class, we also hypothesize the shape of the cost curves. Average costs per volume of water treated will vary with (a) source water quality, (b) flow rate, and (c) target water quality. We expect that for a given type of source water quality, average costs per cubic meter depend on economies of scale. For a *given* source water quality, Fig. 3.1 below shows the hypothesized (theoretical) average costs as a function of the flow rate for different treatment classes. This graph assumes that contaminants are additively separable and linear.

In reality, that assumption of linearity and additive separability would not hold as some technologies can have an overlap in their functions. For example, technologies that can remove suspended solids (Class 2) can also remove some pathogens (Class 3) and possibly some DBP precursors (Class 4), if used in conjunction with coagulation.

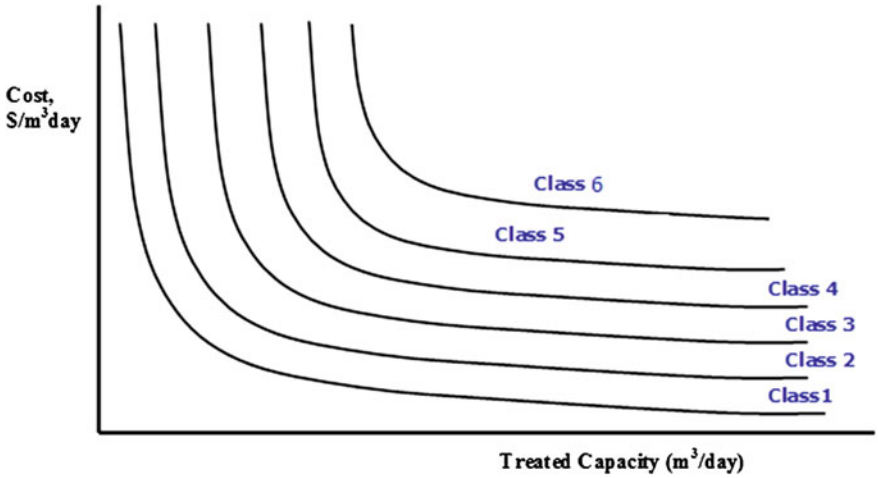

Fig. 3.1 Hypothetical costs curves and scale of treated drinking water

Nevertheless, it might be useful to assess the *cost differentials* between some of the abovementioned treatment classes, and the extent to which nonlinearities might indicate that it would be better to aim at a higher treatment class that happens to have lower average costs per cubic meter even if water quality regulations require just disinfection and no additional removal of contaminants. There is also a further nonlinearity already implicit in Fig. 3.1, namely economies of returns to scale, which suggests that for some smaller communities it might make economic sense to consider a somewhat larger plant scale in the expectation of a future growth in water demand, or consider an amalgamation of two or more small communities to be supplied by a single but larger treatment plant.

It would be interesting to find the average costs per unit of volume of water of the broad water treatment classes and find any nonlinearity in costs where the actual average costs curves may not conform to the hypothetical graph in Fig. 3.1, but in fact exhibit discrete "jumps," indicating the presence of nonlinearities in costs and complementarities in contaminants removed.

3.3 Projected Costs: Ultra Violet, Micro Filtration—Ultra Filtration (MF-UF), High Rate Treatment and Clarification (HRC), and Ozonation

In this section, we present four technologies that may be suitable for small systems: High rate treatment and clarification, UV, MF-UF, Advanced Oxidation Processes (based on UV), Reverse Osmosis-Nano Filtration (RO-NF) and ozonation, although RO is treated in greater detail in Chap. 4 to set the stage for the econometric estimation of "breakeven" and other prices in Chap. 5.

The raw data for costs for different flow rates were obtained from the actual manufacturers (see footnotes for details). For all estimated models, we find average costs per volume of treated water, where the costs are (1) capital costs, amortized (by straight-line depreciation) over a 20 year period, and (2) O&M costs, that include labor, materials, and energy costs for given flow rates.

In the case of surface water, UV-based technologies would most likely require that source water be pretreated using a filtration or sediment removal process before being disinfected by UV. For communities that are concerned about pesticides and other micro-pollutants, advanced oxidation processes (AOPs) may be worth considering. AOPs may not be practical for small systems, but with the implementation of new regulations on drinking water quality in the future, it may be worthwhile for small systems to include UV-oxidation-based treatment technologies in their menu of possible technology options. We note that there are some small communities that are already using AOPs for surface water treatment and also for groundwater remediation, even at a small scale.[2]

We briefly describe each technology in Table 3.3[3] and illustrate the statistically modeled costs associated with each of them thereafter. All the technologies considered here produce municipal standard drinking water, and most assume that the raw source water is surface water, which is easily contaminated by animals and/or human activity.

We use the nonlinear least squares (NLLS) estimation process since it can capture a wider range of functional forms than the ordinary least squares (OLS) method. Simple linear models may not describe certain data generating processes very well especially if the functional form changes over its domain. For instance, our cost data for UV (see Fig. 3.2) shows that a much better description of the data can be had if a nonlinear approach (solid line) is used instead of a strictly linear one (dashed line). In fact, since most of our data followed the same format as in Fig. 3.2, we used the NLLS method to estimate cost functions for the different classes of technology. The NLLS technique has the added advantage of yielding better estimates when the amount of data is limited.[4]

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[2] Stockton California remediates its groundwater using Trojan UV Environmental Contamination Treatment, an AOP; their flow rate is 1,100 cubic meters per day.

[3] This information was collected from a number of companies that produce each technology.

[4] Of course if we make the error term multiplicative, then we could estimate the model by simply taking logarithms. But then we would have to assume that the logarithm of the error term is normally distributed. There is no justification for such an assumption. Here we follow the practice of standard statistical models in which the error term is always additive, representing all omitted variables. The objective is to estimate economies of scale given by the estimated exponent in the nonlinear least squares model. This estimated exponent is the constant elasticity, as is well known.

Table 3.3 Treatment Technologies

Technology	Description	Treatment class
High rate treatment and clarification[a]	• Consists of a clarification system (Actiflo) and filtration system (Dusenflo Mixed Bed Filters)	Class 2
	• Reduces turbidity, color, suspended solids, algae, taste and odor (T&O), metals and total organic carbon	
	• The resulting filtered water from the Dusenflo gravity filter can contain little or no Giardia and Cryptosporidium cysts	
	• MINIMUM PLANT SIZE: 473 m^3/day	
UV System[b]	• Utilizes the ability of ultra violet rays to deactivate microorganisms	Class 3
	• This system on its own is chemical free and produces no disinfection byproducts	
	• However, it can also be used in conjunction with other treatment processes forming a “multi-barrier” approach for treating water for drinking purposes	
	• UV will inactivate bacteria, viruses and protozoa, including Giardia and Cryptosporidium with a dose of 40 mJ/cm^2	
	• We assume some filtration system to remove sediments (e.g. sand filtration) would be required and is included in the cost	
	• MINIMUM PLANT SIZE: 200 m^3/day	
MF-UF[c]	• Micro filtration and ultra filtration involves separating water from organic and inorganic matter contained in the water by forcing it through a micro porous membrane	Class 3
	• Pore sizes in microfiltration membranes are 0.1 to 10 microns thick while ultra filtration membranes are between 0.001 and 0.1 microns • Microfiltration will remove Giardia and Cryptosporidium cysts, bacteria, and some viruses; however not all viruses can be removed via this process. • Microfiltration is also used in sterilization of beverages and pharmaceuticals, clearing of fruit juices, wine and beer, separation of oil-water emulsions and pre-treatment of water for Nano-filtration and reverse osmosis	
	• Ultra filtration removes all viruses, bacteria and suspended solids between 0.001 and 0.1 μm. Ultra filtration is used in paint treatment, oil-water emulsion separations, the food industry and textile industry	
	• MINIMUM PLANT SIZE: 379 m^3/day	

(continued)

Table 3.3 (continued)

Technology	Description	Treatment class
Ozonation[d]	• Ozonation systems utilize the ability of ozone to inactivate microorganisms through oxidation	Class 4
	• The system consists of an ozone pretreatment unit, a BioSand filter and a BioCarbon filter	
	• The roughing filtration system removes suspended solids and coliforms as well as some Cryptosporidium	
	• The BioSand Filter is used to treat parasites, color, cysts, manganese, mercury, iron and turbidity while the BioCarbon Filter treats dissolved organic carbon, tannins, pesticides, iron, bacteria, color and odors	
	• MINIMUM PLANT SIZE: 11.4 m^3/day	
Advanced Oxidation (based on UV)	• A UV-oxidation process designed to provide disinfection and Taste &Odor treatment; it destroys Geosmin and 2-methylisoborneol	Class 5
	• Also removes pharmaceutical, personal care products, pesticides and trace contaminants	
	• System consists of a UV reactor, H_2O_2 dosage and storage system. We assume some filtration system to remove sediments (e.g. sand filtration) would be required and is included in the cost	
	• MINIMUM PLANT SIZE: 818 m^3/day	
RO-NF[e]	• Removes all suspended solids, viruses, bacteria, pathogens and all forms of biological contaminants	Class 6
	• Removes mono and multivalent ions, salts and organics	
	• Essentially passes only pure water. Smallest pore size for membranes to date	
	• MINIMUM PLANT SIZE: 1893 m^3/day	

[a] Produced by Veolia Water Solutions & Technologies in France under subsidiaries John Meunier and Kruger USA
[b] Produced by Trojan Technologies in Canada
[c] MF and UF information obtained from Koch Membrane Systems and Lenntech Water Treatment Solutions
[d] Information for ozonation obtained from Mainstream Water Solutions Inc.
[e] A thorough description can be obtained from Koch Membrane Systems

Table 3.4 shows the estimated cost functions for the various technologies[5] described above with the functional form $y_i = \beta_1 X_i^{\beta_2} + \varepsilon_i$ where y_i is the average

[5] Data were obtained from John Meunier Inc (for HRC), Kruger USA (Actifloc), Trojan Technologies (UV), KOCH Membrane (MF-UF), US Filter, Memcor (MF-UF) and Mainstream Water Solutions Inc. Data for HRC and MF-UF were in US dollars and were converted to Canadian dollars. However, all data were converted to a base year (2008) in Canadian dollars for proper comparison.

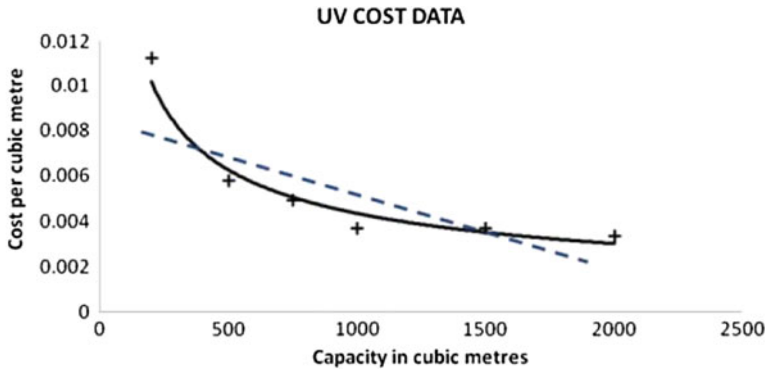

Fig. 3.2 Ultraviolet (UV) Linear Versus Nonlinear Estimation of Cost Data

Table 3.4 Estimated average cost functions for High Rate Clarification & Filtration (HRC), UV, MF-UF and Ozonation in 2008 CDN dollars

Disinfection technology	Average cost function	Predicted cost per cubic meter based on plant with daily capacity		
		100 m^3	200 m^3	500 m^3
HRC	$y = 0.3226x^{-0.2503}$	0.10	0.09	0.07
UV	$y = 0.2653x^{-0.6003}$	0.07	0.06	0.06
MF-UF	$y = 0.4171x^{-0.3048}$	0.10	0.08	0.06
Ozonation	$y = 2.2107x^{-0.381}$	0.38	0.29	0.21

cost per cubic meter, defined as capital plus O&M, X_i is the flow rate in cubic meters and ε_i is the error term, which satisfies the standard Gaussian assumptions.

Details of the NLLS regressions and model fit statistics are given in Appendix A. The estimations provided in Table 3.4 above are based on disinfection for the particular technology only and do not take into account the additional cost of residual chlorine for the distribution system, which is required in the US and Canada. We assume that this additional cost would be the same for all the technologies listed above in Table 3.4, and it was, therefore, left out. In any case for any *actual plant,* there will be many plant-specific costs that the consulting engineers will need to take into account. Therefore, the costs given by the cost models should be viewed as the first approximation to costs; costs of specific water treatment plants are likely to vary.

From Table 3.4 we can observe that both the MF-UF and High Rate Clarification and Filtration (HRC) drinking water treatment can cost on average 10 cents per cubic meter for a 100 m^3 size plant. For surface waters, UV seems to be cheaper than HRC, but direct comparison could be misleading, as a lot of location-specific factors need to be taken into account. (Examples of location-specific factors would be the quality of source water, the presence of color or turbidity, etc.) For UV, some

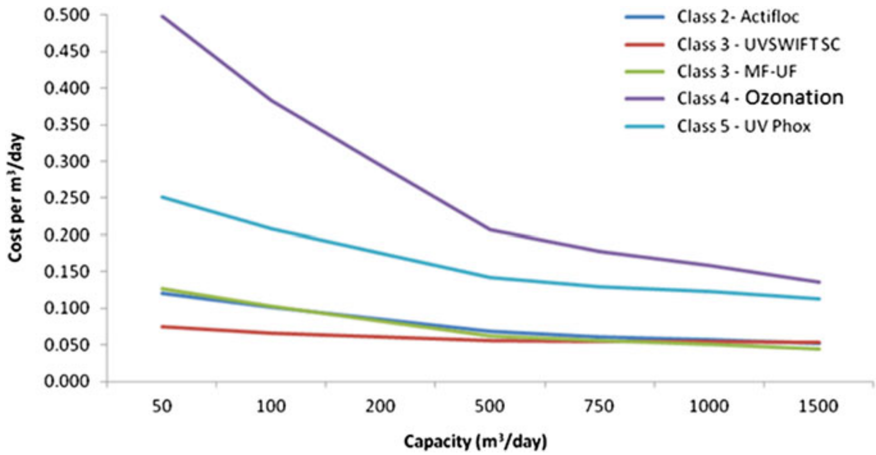

Fig. 3.3 Estimated cost curves: Class 2 for HRC, Class 3 for UV and MF-UF, Class 4 for Ozonation and Class 5 for a UV-based AOP

additional costs must be added for suspended solid removal, such as sand filtration, which could add up to 5 cents per cubic meter, and has been included in Table 3.4 and in Fig. 3.3. Ozonation seems to be the most expensive, but of course it can remove more contaminants and goes beyond disinfection. Perhaps this jump in the classes is a *nonlinear feature,* and therefore the cost per cubic meter increases by an anomalous amount from Class 3 to Class 4.

Ozone treatment plants[6] have expanded rapidly in small systems across Saskatchewan and Manitoba in Canada, mostly for surface water sources. By one count, there were about 30 small ozone plants in operation (at the end of 2010). Compared to a UV-based treatment plant, ozonation is more expensive, but nevertheless it is proving to be attractive to a number of smaller communities.

3.4 Class 5 Treatment Technologies

UV-based advanced oxidation process (AOP) is classified as a Class 5 treatment technology in Table 3.1. Hydrogen peroxide absorbs UV light in order to form free hydroxyl radicals, which aid in breaking down contaminants. A combination of UV-photolysis and UV–Oxidation is therefore used in the treatment process. In Table 3.5 we present the estimated NLLS average cost function for such an AOP.

[6] These Ozone treatment plants were supplied by Mainstream water solutions Inc. Their brand name was SCOR.

Table 3.5 Estimated average cost function for UV-Based AOP in 2008 CDN dollars

Disinfection Technology	Average Cost function	Predicted cost per cubic meter in Can \$ based on plant with daily capacity in m^3		
		100 m^3	200 m^3	500 m^3
UV-Based AOP	$y = 0.7576x^{-0.3394}$	0.21	0.18	0.14

Details of the NLLS estimate are shown in Appendix B. We included an additional cost for filtration for surface waters for this AOP of 5 cents per cubic meter in the predicted costs in Table 3.5. We estimate that Class 5 treatment can cost \$0.21 per cubic meter for a small plant with a daily capacity of 100 m^3. Note that our statistical modeling estimation, based on data supplied by manufacturers, indicates that this Advanced Oxidation Process is cheaper than ozonation and will remove a number of micro-pollutants (see description in Table 3.3). When plant-specific costs are taken into account, our information indicates that a representative plant at a scale of 3800 cubic meters per day would cost around \$0.45 per cubic meter (in 2008 Canadian dollars).[7]

We hasten to add that our cost estimation models yield what we can call “first approximation costs” and what is the most appropriate technology will depend on site-specific (i.e. the particular location) factors. It is best left to the consulting engineers to do a thorough cost estimation for specific sites.

3.5 Reverse Osmosis and Nanofiltration (Class 6)

Reverse Osmosis and Nano Filtration, which can also remove salinity, is classified as Class 6. Dore (2005) shows that for a flow rate of 5,000 cubic meters per day, the cost of producing drinking water was US \$0.50 per cubic meter per day in 2005. In a later article, Fritzmann et al. (2007) put the costs at actual desalination plants to be between US\$0.48 and \$0.53 cents. Finally, in a comprehensive review of the cost of desalination literature, Karagiannis and Soldatos (2008) show that for capacities between 500 and 1,000 m^3, RO costs range from US\$0.75 cents to \$3.93 per m^3 per day. For capacities less than 1000 m^3, they find that the costs range from US \$2.22 to as much as \$19 per m^3 per day. All authors mentioned here recognize the importance of economies of scale in the determination of unit costs. To some extent, RO with granular activated carbon is a “state-of-the-art” technology, mainly suitable for large systems, and so RO and other such technologies are considered in greater detail in Chap. 4.

We can also compare the above cost data with the costs of a Point-of-use (POU) Reverse osmosis system. POU costs range from 2.5 to 5 cents per liter or \$25 to \$50

[7] Personal communication from Mr. Morris McCormick, Drinking water treatment plant, City of Cornwall, Ontario.

per cubic meter. These are obviously expensive technologies and possibly not suitable for small water systems.[8]

3.6 Examples of Actual Costs of a Few Existing Plants

In this section, we present costs and flow rates at some existing water treatment plants in select small communities in British Columbia (BC), Canada. As before, the costs are made up as follows: (1) capital costs, amortized over a 20-year period, and (2) O&M costs, that include labor, materials, and energy costs for given flow rates. Some of these plants are managed by private corporations as operators, and therefore include their profit markup. The cost information was obtained from the managers of these water treatment plants.

Table 3.6 shows the Class and flow rate as well as its associated average operating cost per cubic meter per day. The largest flow rate plant analyzed here produces the least expensive drinking water (compared to other facilities in the same province) at $0.39 per cubic meter per day. The plant that provides the most costly drinking water also has one of the lowest flow rates.

Using the actual data from these select small systems in BC, we estimate various cost functions for different classes of technology. Note that for Class 1, for some of these communities, the costs reflect (a) profit markup for private sector management, (b) higher transportation costs of hazardous materials such as chlorine and (c) higher transportation costs due to remoteness. These privately managed water systems have costs that include a 100 percent markup on labor costs. We estimated the average cost functions based on the NLLS estimation procedure (see Table 3.7).

Table 3.6 Some examples of existing small water treatment facilities in BC for 2008

Class	Treatment used	Scale (m^3/day)	Operating Cost per year ($)	Unit Operating cost ($ per m^3/day)
1	Chlorination only	92	41,128	1.23
1	Chlorination only	50	23, 536	1.28
1	Chlorination only	126	40,496	0.88
1	Chlorination only	38	30,202	2.16
1	Chlorination only	778	111,641	0.39
2	Chlorination plus removal of suspended solids	46	46,247	2.72
4	Chlorination plus removal of suspended solids, protozoa and dissolved organic content	640	100,000	0.59

[8] In this chapter we do not pursue these costs; POU systems are being investigated in a separate research project.

Table 3.7 Examples of estimated average cost functions for BC small systems in 2008 CDN dollars for three capacity levels

Water treatment classification	Average cost function	Predicted cost per cubic meter based on plant with daily capacity		
		100 m^3	200 m^3	500 m^3
Class 1	$y = 19.343x^{-0.6428}$	1.00	0.64	0.36
Class 2	$y = 25.537x^{-0.5998}$	1.61	1.06	0.61
Class 4	$y = 375.873x^{-1.000}$	3.76	1.88	0.75

Details of the NLLS estimation are shown in Appendix C. Costs shown above for Class 1 are operating costs for treatment only. Class 1 plants with a daily flow rate of 100 m^3 can produce drinking water at an average cost of \$1.00, while the cost is almost quadrupled for a similar sized plant producing Class 4 drinking water on an island off the coast of British Columbia.

3.7 Summing up and Tentative Conclusions

We can now show, in Figs. 3.3 and 3.4, that with the estimated cost functions, we can reproduce an actual set of cost functions that can then be compared to the hypothetical Fig. 3.1. Figure 3.3 shows the estimated cost curves based on manufacturers' rated costs, while Fig. 3.4 shows the estimated cost curves based on a sample of small systems in BC.

Figure 3.3 indicates that ozone technology, a Class 4 water treatment, is more expensive than the Class 3 (UV and MF-UF) and Class 2 (HRC) treatment types.

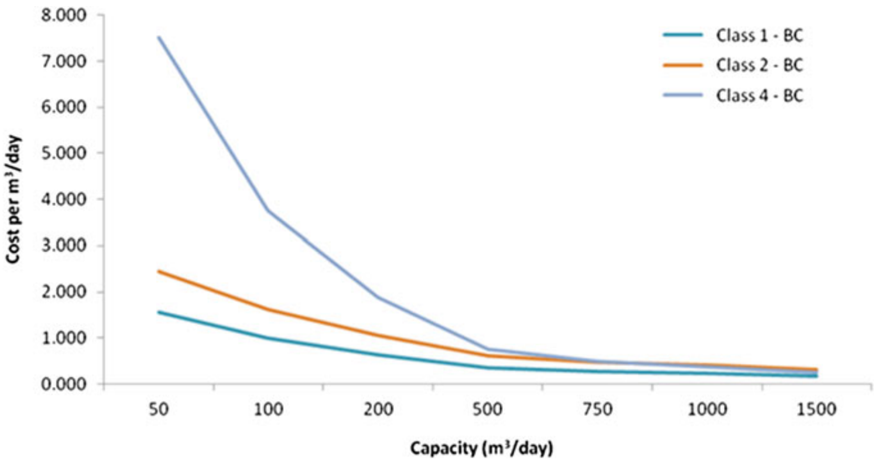

Fig. 3.4 Estimated cost curves: Classes 1, 2 and 4 for BC Small Systems

Class 3 treatments MF-UF and UV seem to be cheaper than HRC for plants which produce less than 100 m^3 of water per day and all the way up to 500 m^3/day, even though HRC is a Class 2 water treatment process. But in general Fig. 3.4 suggests that the higher the Class of water treatment is, the higher will be the average costs per cubic meter for the sample of small systems in BC.

We observe that the average cost per cubic meter of the statistically estimated equations given above do not conform exactly to the hypothetical Fig. 3.1, but exhibit the nonlinearities that we expected. Another nonlinearity may be the cost of moving from one technology to another, especially when there has been a long-term commitment to a particular technology.

It is possible that older small systems continue to use higher cost older technologies as there is no incentive to modernize in the public sector. In other words, there are technologies currently available in the market that can provide higher contaminant removal at a much lower cost per cubic meter. Hence, we find that a technology, which can provide Class 3 and 4 water treatment, shows lower average cost per cubic meter than a small system, which is only providing Class 1 and 2 water treatments. Another possible reason is that there are site-specific costs that can contribute to the gap in the costs functions between technology and actual existing systems that are in the same class. For example, many of the small systems in BC mentioned above have higher transportation costs due to remoteness and the handling of hazardous materials such as chlorine. However, site-specific costs alone cannot account for this very large gap. We observe that some treatment classes at lower flow rates dominate in terms of cost-effectiveness. Class 3 MF-UF and UV provide water treatment at a much lower cost per cubic meter than BC small systems Classes 1 and 2 between output flow rates of 50 to 200 m^3 per day; but at higher flow rates this gap tends to decrease. Finally, the cost per unit for these existing BC small systems is high compared to the rated costs because the systems are privately owned and costs include a markup for profit.

Before we summarize the conclusions, we need to distinguish between systems that use groundwater as the source and systems that use surface water as the source. Most of the above analysis is concerned with surface water as the source for water treatment plants.

Based on Figs. 3.3 and 3.4 and the results presented in the previous sections, we provide the following tentative conclusions:

1. The estimated cost curves show that small systems could achieve a higher removal of contaminants at a lower cost than their currently used technology;
2. A small publicly owned system could get Class 2 and 3 water treatment if they use HRC or MF-UF for about 9 to 11 cents respectively, provided the flow rate is 100 m^3 per day;
3. For systems using surface water, UV appears to be the least expensive for small systems at only 7 cents[9] per m^3 for a plant with capacity of 100 m^3 for Class 3, which shows that the competitive advantage remains even when costs of

[9] Includes 5 cents for sand filtration or sediment removal.

sediment removal are included. We would argue that where primary disinfection is absolutely necessary, UV would compare favorably with chlorine for primary disinfection. Of course in North America, for the distribution system the law requires chlorine residual, and perhaps that is why many small systems continue to rely on chlorine as a primary disinfection for surface water systems. The concern over disinfection byproducts (DBPs) might tip the scale in favor of UV for primary disinfection. But again site-specific considerations need to be taken into account. Furthermore, when the source water is groundwater, which is otherwise free of contaminants, the only cost is the cost of residual chlorine for the distribution system. In this case, chlorine may be cheaper than UV.

4. If a community is concerned with the removal of micro-pollutants, then a UV-based Advanced Oxidation Process would be cheaper than ozonation, provided the flow rate is not too small. (For example, the City of Cornwall in Canada uses AOP for 2 months of the year for taste and odor issues.)
5. Our results indicate that ozonation is competitive (2008 CDN $), and so there are number of ozonation plants in Saskatchewan and Manitoba. We estimate that at the beginning of 2011, there were 30 small systems using this technology in the two provinces (Table 3.8).
6. In general, manufacturers' rated costs tend to be lower than actual plant-level average costs as they do not include some plant-specific costs, such as higher labor, energy, and transportation costs due to remoteness from large urban areas (Table 3.9).
7. It should be noted that some of the estimations are based on limited data. Needless to add that the costs estimates cannot be treated for predictive purposes, as all useful predicted costs must also take into account a number of location-specific costs (Table 3.10).

Our general conclusion is that while any specific water treatment facility will need to take account of raw source water quality, the *actual* target quality for small systems seems to be to meet only the *minimum regulatory requirements*. Our results show that for surface water, unless the raw water is high in color and in turbidity, a UV-based plant would be economical and cost-effective even when the additional cost of sediment removal is added. This conclusion is especially true for small plants producing less than 100 cubic meters per day. Such a plant could obtain the same or better quality water with UV for less than 8 cents per cubic meter per day. Our finding of the cost-effectiveness of UV is in agreement with USEPA (1996), Gadgil (1998) and Parrotta and Bekdash (1998).

Appendix A

Estimation results for Table 3.4 based on the model: $y_i = \beta_1 X_i^{\beta_2} + \varepsilon_i$ where y_i is the average cost per cubic meter, X_i is the capacity in cubic meters and ε_i is the error term, which satisfies the standard Gaussian assumptions

Table 3.8 Estimated regression coefficients

Coefficients	HRC	UV	MF-UF	Ozonation
β_1	0.3226	0.2653	0.4171	2.2107
	(0.077)	(0.025)	(0.060)	(0.000)
β_2	−0.2503	−0.6003	−0.3048	−0.381
	(0.027)	(0.000)	(0.003)	(0.000)
R^2	0.946	0.975	0.853	0.999
$\bar{R}^2$	0.919	0.968	0.829	0.999
S.E. β_1	0.115	0.076	0.180	0.065
S.E. β_2	0.042	0.048	0.061	0.009
No. of Observations	4	6	8	4

p-values in parentheses

Appendix B

Estimation results for Table 3.5 based on the model: $y_i = \beta_1 X_i^{\beta_2} + \varepsilon_i$ where y_i is the average cost per cubic meter, X_i is the capacity in cubic meters and ε_i is the error term, which satisfies the standard Gaussian assumptions

Table 3.9 Estimated regression coefficients

Coefficients	UV-based AOP
β_1	0.7576
	(0.4357)
β_2	−0.3394
	(0.1397)
R^2	0.759
$\bar{R}^2$	0.638
S.E. β_1	0.784
S.E.β_2	0.142
No of Observations	4

p-values in parenthesis

Appendix C

Estimation results for Table 3.7 based on the model: $y_i = \beta_1 X_i^{\beta_2} + \varepsilon_i$ where y_i is the average cost per cubic meter, X_i is the capacity in cubic meters and ε_i is the error term, which satisfies the standard Gaussian assumptions

Table 3.10 Estimated regression coefficients

Coefficients	Class 1	Class 2	Class 4
β_1	19.343	25.537	375.873
	(0.015)	(0.494)	(0.000)
β_2	−0.6428	−0.5998	−1.000
	(0.000)	(0.162)	(0.000)
R^2	0.773	0.413	0.999
$\bar{R}^2$	0.765	0.266	0.999
S.E[a]. β_1	7.456	33.939	0.000
S.E. β_2	0.096	0.351	0.000
No of Observations	29	6	4

[a] Standard Error

Appendix D

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Chapter 4
Reverse Osmosis and Other Treatment Technologies

4.1 Introduction

In the previous chapter we classified water treatment into six classes. We also stated that reverse osmosis and other advanced treatment processes produce a higher quality of treated drinking water. This chapter is a description of Reverse Osmosis (RO) and other similar treatment processes. In Chap. 5, we use data from 36 RO desalination plants as our sample for estimating breakeven prices and "Shadow Ramsey prices," which are prices adjusted to reflect marginal social opportunity costs; these prices can be considered long-run marginal cost prices.

In order to estimate the prices in the next chapter, we need *Total Cost* and *Plant Size* information on water utilities which are currently using RO treatment technology, which is probably the least-cost, state-of-the-art technology for producing high-quality drinking water. Accordingly, this chapter focuses on such technologies available to water utilities today. It so happens that RO can also desalinate seawater and other brackish water and, therefore, the processes were developed for water scarce areas that could use seawater to produce drinking water.

This chapter is organized as follows. The following section is a general introduction to desalination treatment technologies and how these have grown over time. Section 4.3 gives details of desalination processes of which RO is the most prominent. Section 4.4 covers the relative costs of desalination as a function of scale. It includes a description of the 36 RO treatment plants worldwide, which form our sample of plants that are the subject of econometric estimation in Chap. 5. The final section draws the conclusions.

M.H. Dore, *Global Drinking Water Management and Conservation*,
Springer Water, DOI 10.1007/978-3-319-11032-5_4

4.2 Water Desalination Technology in Application

Water desalination is a water treatment process that removes salts and other minerals from water. Desalination or desalting can be done in a number of ways, but the result is always the same: freshwater is produced from brackish (up to 10 g of minerals/L) or seawater (up to 50 g of minerals/L). This is equivalent to saying that about 3.5 percent of seawater is made up of dissolved salts. According to WHO, RO will remove 99 percent of all (large molecule) contaminants. But typically, RO is accompanied by activated carbon filtration, which will remove any remaining traces of contaminants. The resulting treated water is then distributed to consumers for drinking and other purposes (Fig. 4.1).

On June 30, 2011, there were 15,988 desalination plants worldwide, and the total global capacity of all plants in operation was 66.5 million m^3/day, or approximately 17.6 billion US gallons per day (see Fig. 4.2). Saudi Arabia has 6.5 million m^3/day of installed capacity.

In the US, there are over 2,000 desalination plants over 32 states, amounting to 6 million m^3/day of installed capacity. Of this total, 324 have an installed capacity of over 95 m^3/day (or 25,000 gallons per day). Their cumulative capacity is over 4.5 million m^3/day, and 94 percent of these are drinking water treatment plants; the rest are wastewater treatment plants.

As shown by Dore (2005), desalination capital and operating costs have been decreasing. The decrease in costs is partly attributable to tax breaks for R&D in the US. The year 1952, when the United States federal government passed the Saline

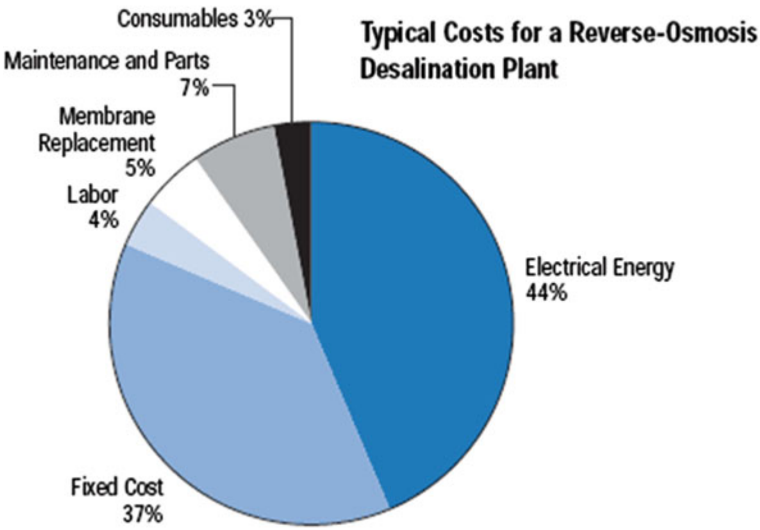

Fig. 4.1 Distribution of the cost components of a typical reverse osmosis desalination plant. (Wilf 2004)

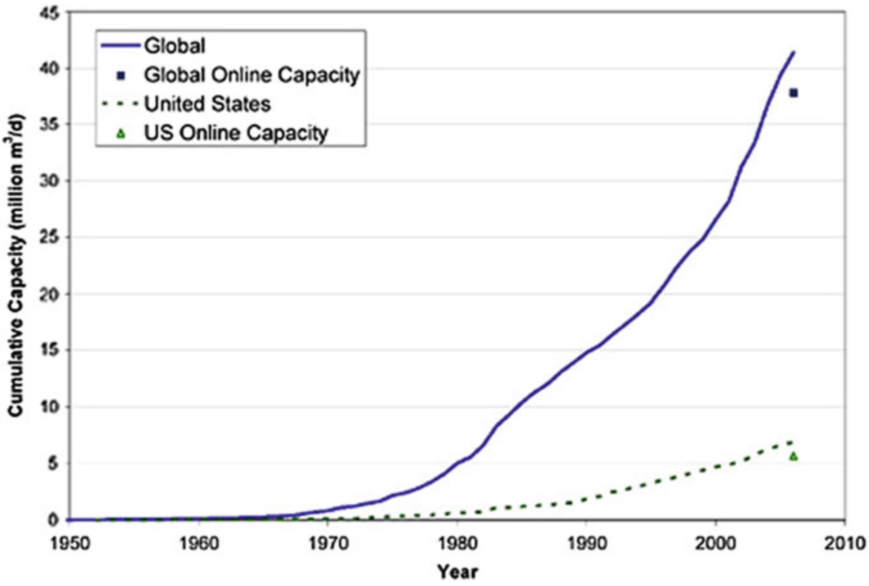

Fig. 4.2 Cumulative capacity of installed desalination plants in the US and worldwide from 1950 to 2006 (National Research Council 2008)

Water Act, marked the beginning of a long line of legislations that were designed to fund desalination research and development. By the mid-1960s, the US had built 45 percent of all the desalination plants in operation around the world. In 1964, the Water Resources Research Act was introduced in the US, which provided funding for desalination R&D. During the mid- to late-1960s, much research was done on developing membranes and distillation technology. The technology developed during this period was made freely available worldwide, through workshops and published reports. This easy access to the technology contributed to the decrease in costs of desalination. The US continued to lead in desalination technology in the 1960s and 1970s, followed by Europe and Japan. In 1973, the oil embargo increased distillation costs and the need for more energy research caused a rapid decrease in ongoing desalination R&D. By 1974, RO had become commercialized, reducing the need for federal support. In 1976 and 1977, the western United States experienced a drought, which increased the interest in desalination technology. This led to the Water Research and Conversion Act of 1977, with desalination research focusing on RO. From 1953 to 1982, the US federal government spent over $1 billion (in 1999 dollars) on desalination research (US Congress, Office of Technology Assessment 1988). It is generally accepted that this government involvement was responsible for the development of RO (Dore 2005). In 1996, after a decade of limited government support to desalination research, the Water Desalination Act was enacted. The Act reflected a renewed interest in desalination technology and aimed at developing more cost-effective and efficient technologies.

Dore (2005) estimated that the federal government's financial assistance was in the range of \$5,000–\$125,000, with the average grant amounting to \$60,000 per research or study activity. In 2000, 15 financial assistance agreements were awarded.

Figure 4.3 shows how the unit capital costs of installed capacity decrease as the capacity increases. Furthermore, the total unit costs (capital and operating costs per cubic meter of desalinated water) have been decreasing, while the cost of obtaining and treating water from conventional sources has been increasing (see Fig. 4.1 for the distribution of the cost components for RO). The reasons for the increase are summarized in Table 4.1. In fact, water quality standards have become more stringent in many countries, requiring increased levels of treatment. Also, conventionally treated water costs more because of an increased demand for water, resulting in the development of more expensive conventional sources, since the less expensive sources have already been used.

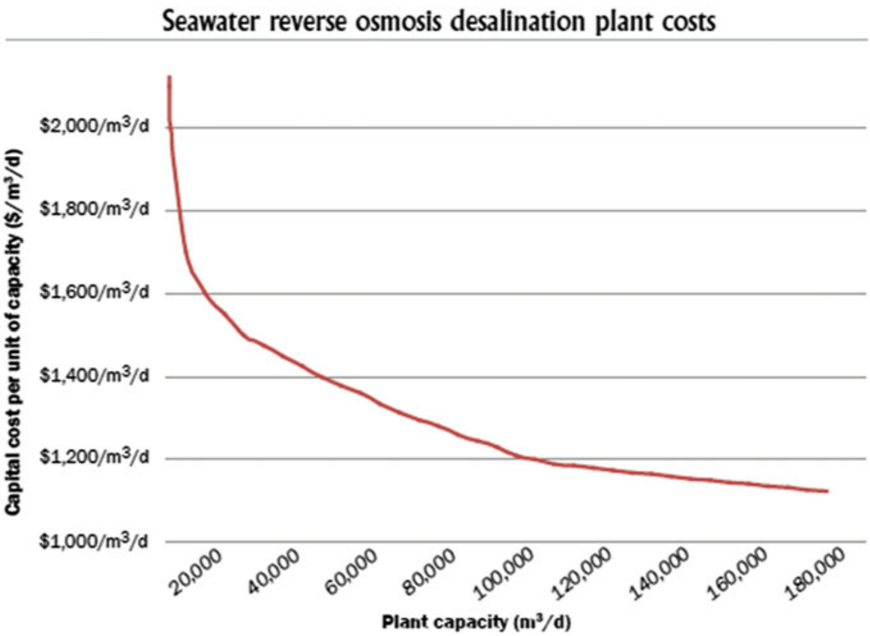

Fig. 4.3 Economies of scale in the capital costs of seawater reverse osmosis desalination plants (Global Water Intelligence 2009)

Table 4.1 Reasons for the rising costs of traditional water sources

Increasing levels of treatment being required to meet more stringent water quality standards
Increasing demand for freshwater
Decreasing supplies of freshwater
Increasing costs of maintaining existing water supplies in a fresh state
Alteration of existing pricing schedules to reflect true costs of provision

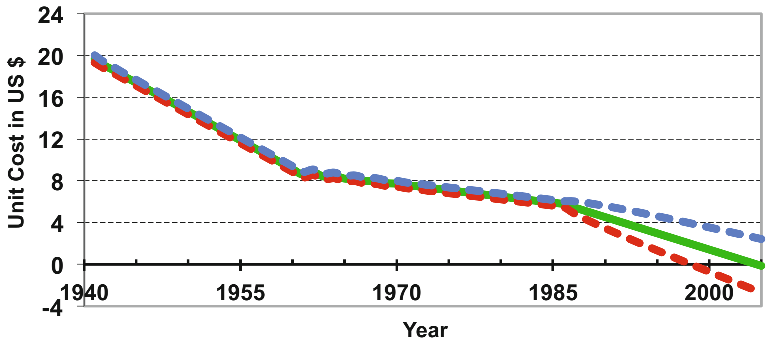

Fig. 4.4 Integrated moving average model forecast (Dore 2005)

Based on the cost information from US plants that have been using RO to produce drinking water through desalination, Dore used an autoregressive integrated moving average model (ARIMA) to forecast the change in real seawater desalination unit costs. The obtained model is shown graphically in Fig. 4.4.

Figure 4.4 shows a steady decrease in real seawater desalination unit costs from 1940 to 1988, as well as a further decrease in forecasted costs up to the year 2005. The blue line represents the upper 95 percent confidence interval of the forecast and the red line represents the 95 percent lower confidence interval of the forecast. The green line is the predicted value. From Fig. 4.4 we see that real unit costs of desalination are expected to continue their downward trend. For the year 2000 the real unit cost of seawater desalination was expected to lie within \$0.00 to \$3.54 per 1,000 gallons (3.79 m^3). For the year 2005 this cost was expected to fall even further and lie within \$0.00 to \$2.42 per 1,000 gallons (Dore 2005).

Based on this forecast and the fact that the cost of obtaining and treating water from conventional sources has been increasing, we can expect that desalination costs will soon be competitive with those of conventional water treatment processes. This suggests that desalination could become an important source of drinking water even when the feed water does not require desalting.

4.3 Desalination Processes

Desalination processes can be divided into two categories: (a) thermal methods, which involve heating water to its boiling point to produce water vapor and (b) membrane processes, which employ a membrane to move either water or salt to induce two zones of differing concentrations to produce freshwater. Desalination facilities use one of five basic technologies to extract potable water from sea and brackish water. The five technologies include RO, distillation, electrodialysis, ion exchange, and freeze desalination. These technologies are classified as thermal or membrane processes in Table 4.2.

Table 4.2 Desalination technologies—classified

Thermal methods	Membrane processes
Distillation	Electrodialysis
Freeze desalination	Reverse osmosis
	Ion exchange

Each technology has its own set of advantages and disadvantages depending on the permitted plant size, concentration of organic and inorganic material in treated water, desired quality of water produced, and the availability of energy and chemicals to treat the water and their costs.

4.3.1 Reverse Osmosis

Reverse osmosis is a three-stage process represented in Fig. 4.5. In stage one, water is pretreated and screened to remove particles that would clog the RO membranes. In stage two, water is forced under high pressure through several semi-permeable membranes. The membranes restrict salts and other contaminants while allowing water molecules to pass through to the final stage. The final stage is referred to as the posttreatment stage. During the posttreatment stage freshwater may have its pH level adjusted or be combined with chlorine or various chemicals for disinfection and transportation. The end result is high-quality water fit for human consumption with approximately 10–500 parts per million of dissolved solids.

Recent advancements in membrane technology have allowed the cost of purifying water to drop substantially while at the same time increasing the quality of the water. In the past, RO plant operators had difficulty in keeping the membrane surface clean, particularly when treating seawater, surface water, or wastewater. Often the highly contaminated water would clog the membranes and reduce the flow capacity of the plant. As a result, higher operating costs due to cleaning chemicals, down time, and increased labor cost to clean the membranes made desalination a costly method of water purification.

To prevent the RO membranes from clogging, an *Ultra-filtration* pretreatment membrane has been developed. The ultra-filtration membrane consists of

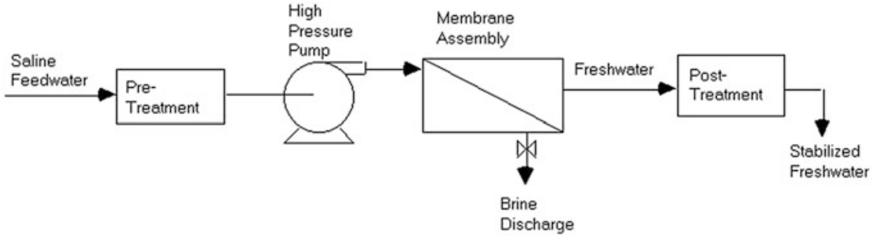

Fig. 4.5 Flow diagram of a reverse osmosis system (California Coastal Commission 1993)

polysulfone hollow fiber membranes, which are placed asymmetrically with the feed stream. The advantage of the polysulfone membrane is that it is chlorine tolerant. Therefore, as biological growths build up on the membrane, a backwash of water combined with chlorine can quickly sterilize the membrane. In addition, the ultra-filtration membrane has a barrier surface that is capable of removing water-borne pathogens, such as Cryptosporidium, Giardia, and other viruses.

Since the ultra-filtration pretreatment technology can be easily cleaned and prevents waterborne pathogens, the water pressure necessary to force the water through the RO membrane can be substantially reduced. In many cases energy requirements will be as little as 4.69 kWh/m^3 of water to operate depending on the amount of contaminants in the water (Dore 2005). In 2006, the energy needs for RO had fallen to 2.33 kWh/m^3, according to Gilau and Small (2006).

4.3.1.1 Recent Experience with New Desalination Projects in the USA

Two recent examples of RO desalination plants are instructive. First, the desalination plant in Tampa, Florida, is the largest operating seawater facility in North America at 25 MGD, or 94, 635 m^3/day. In the initial planning in the late-1990s this plant was expected to be a cost-effective supply option. However, the Tampa plant, a facility to desalinate heavily brackish estuarine water, encountered technical and economic problems. For example, less freshwater was produced than anticipated; the RO membranes were subject to fouling, and financing issues during construction and startup raised the cost of the freshwater produced. The Tampa project illustrates some of the risks of working with private sector water developers, without an adequate external review and the establishment of sound accountability mechanisms (Carter 2013).

Second, in 1998, north of San Diego in Carlsbad, California, a private joint venture, Poseidon, initiated its effort to build a 50 MGD (189,271 m^3 per day) seawater desalination facility to sell water to San Diego's water system. In November 2009, "Poseidon Carlsbad" project received the necessary permit. In November 2012, the San Diego County Water Authority approved the purchase of the desalinated water for 30 years. In 2012, the project costs were estimated at close to $1 billion, which represents a significant increase from estimates a decade earlier at $270 million; the cost for delivered desalinated water from the plant is estimated at $1,600 per acre-foot, or $1.41 per cubic meter. The plant is expected to complete construction and begin water deliveries in 2016. The participation of the private sector necessitated lengthy negotiations, which took time. While Poseidon owned a prime location site for a desalination facility, the water authority and public were hesitant about the arrangement because of concern over profit-taking by a private entity engaged in the provision of a public service. After more than a decade, this and other concerns (e.g. environmental impacts) were overcome. The Poseidon Carlsbad experience has yielded lessons about the public's expectations for transparency and protections when the private sector is involved in desalination or other aspects of public services and infrastructure (Carter 2013).

4.3.2 Distillation

The second method of desalination is distillation. In the distillation process, raw water is heated and then evaporated to separate dissolved minerals and to kill harmful bacteria. The steam is then condensed and collected. The four most common methods of distillation to produce freshwater for commercial or semi-commercial applications are multistage flash, multiple effect distillation, vapor compression, and solar distillation.

In multistage flash, represented in Fig. 4.6, the raw water is heated and the surrounding pressure is lowered. Since the pressure is lower, the boiling point of water is reduced and the water "flashes" into steam. This process continues over and over again at lower and lower pressures until the waste content becomes too high. In general, the electricity requirements of distillation are considerably higher than the RO process. This higher energy input is indicative of the inefficient conversion of electricity to heat.

In multiple effect distillation, represented in Fig. 4.7, the inefficient exchange of energy to heat is reduced by reusing the heated wastewater. As raw water enters an evaporator it is heated and turned into steam. The wastewater is then used to assist in the heating of the next evaporator. This process continues until the wastewater cools and can no longer provide sufficient heat.

Vapor compression is similar to multiple effect distillation except the inefficient exchange of energy to heat is reduced even further. In vapor compression, represented in Fig. 4.8, the raw water is heated and turned into steam. This steam is then compressed and used to assist in heating the next evaporator. This process continues until the steam is condensed back into water.

Solar distillation offers the lowest operating cost of the distillation methods since the heat energy is provided with little cost by the sun. In solar distillation, represented

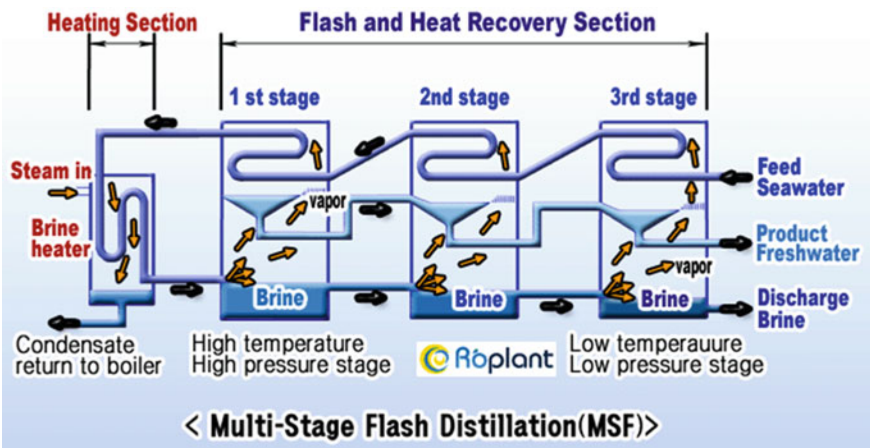

Fig. 4.6 Flow diagram of a multistage flash system (Water Industry Portal 2014)

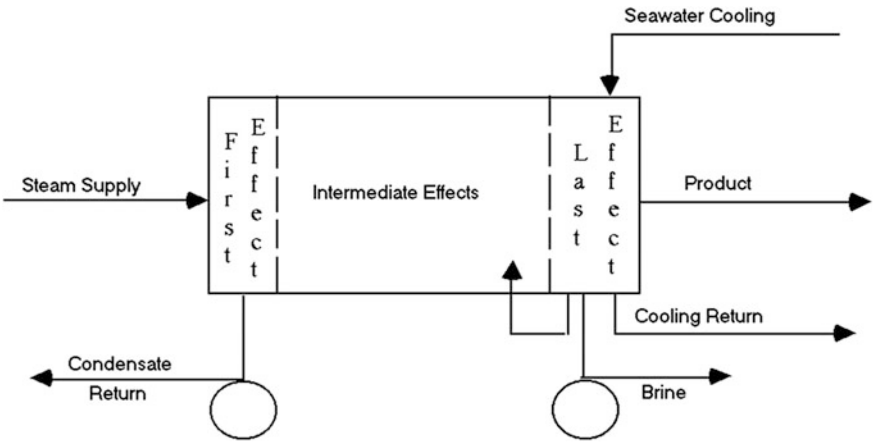

Fig. 4.7 Flow diagram of a multiple effect distillation system (California Coastal Commission 1993)

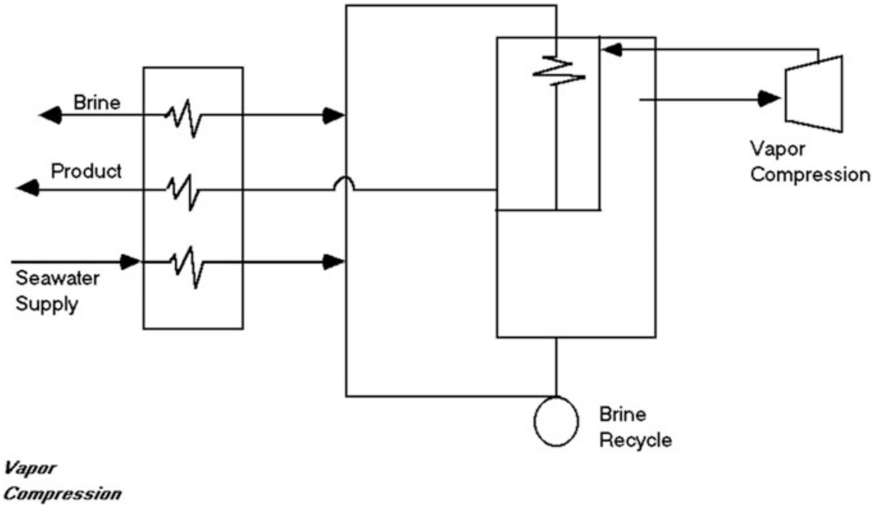

Fig. 4.8 Flow diagram of a vapor compression system (California Coastal Commission 1993)

in Fig. 4.9, solar radiation provides the energy to turn the raw water into steam. Once the water enters a gaseous state it rises and collects at the top of the glass or plastic barrier. The steam then cools, condenses, and is collected as freshwater.

In the past, solar stills produced only 6 l of freshwater per square meter per day of collector surface (Kunze 2001). This would mean that approximately 3.78 million square meters of collector surface would be required to match the modest production capacity of the Sand City six mega-gallon-a-day desalination facility. Since space of this magnitude is rarely available, solar stills are a poor choice for large-scale freshwater production. Even with modern advances in heat recovery and

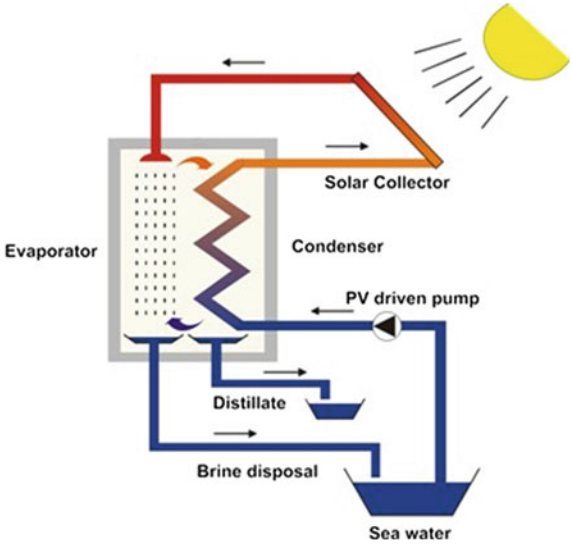

Fig. 4.9 Flow diagram of a solar still (Freeze Desalination 2014)

air mass circulation, the production capacity has only reached 20 l of water per square meter per day of collector surface (Kunze 2001).

Distillation is an energy intensive process compared to RO. In Table 4.3 we see that distillation energy requirements consist of electricity to operate the plant and heat to boil the raw water. When this heat energy is generated by electricity, the electrical power requirements increase 5–10 times their original value. Due to this heavy dependency on heat energy, 50–75 percent of distillation unit costs are made up of heat energy costs (Winter et al. 2002).

4.3.3 *Electrodialysis*

The third method of desalination is called electrodialysis. This involves pumping of brackish water at low pressure between several hundred flat, parallel, ion-permeable membranes that are assembled in a stack. Electrodialysis is used to transport salt ions from one solution through ion-exchange membranes to another solution under

Table 4.3 Energy requirements for reverse osmosis and distillation (California Coastal Commission 1993)

Desalination technology	Electricity requirements
Multistage flash	2.80–5.60 kWh/m^3 plus heat energy
Multiple effect distillation	2.00–4.00 kWh/m^3 plus heat energy
Vapor compression	8.00–12.00 kWh/m^3 plus heat energy
Reverse osmosis—single pass	4.64–8.80 kWh/m^3
Reverse osmosis—double pass	5.20–9.60 kWh/m^3

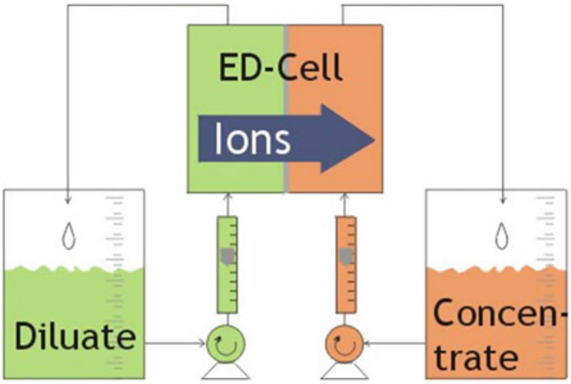

Fig. 4.10 Flow figure of the electrodialysis process (PCA —Polymerchemie Altmeier GmbH 2014)

the influence of an applied electric potential difference. This is done in a configuration called an electrodialysis cell. The cell consists of a feed (dilute) compartment and a concentrate (brine) compartment formed by an anion exchange membrane and a cation exchange membrane placed between two electrodes. Electrodialysis processes are different compared to distillation techniques and other membrane-based processes (such as RO) in that dissolved species are moved away from the feed stream rather than the reverse (Fig. 4.10).

4.3.4 Ion Exchange

The fourth method of desalination is called ion exchange. In this process undesirable ions in the raw water are exchanged for desirable ions as the water passes through granular chemicals called *ion exchange resins*. For example, cation exchange resins are typically used in homes and municipal water treatment plants to remove calcium and magnesium ions in “hard” water, and by industries in the production of ultra-pure water. The higher the concentration of dissolved solids in the raw water, the more often the resins will need to be replaced or regenerated. With rising costs for resins and for disposing of regeneration solutions, ion exchange is now competitive with RO and electrodialysis only in treating relatively dilute solutions containing a few hundred parts per million of dissolved solids.

4.3.5 Freeze Desalination

The fifth method of desalination is called freeze desalination. Freeze desalination is a five-stage process represented in Fig. 4.11. In stage one, raw water is precooked to remove harmful bacteria. In stage two, the water is frozen into slush, where ice crystals are allowed to form. The ice is then removed in stage three and washed in stage four. Once clean, the ice is melted to form fresh drinkable water. While the

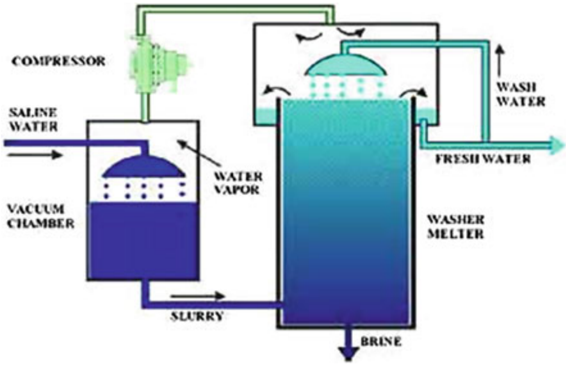

Fig. 4.11 Flow diagram of the freeze desalination process (Freeze Desalination 2014)

feasibility of freeze desalination has been demonstrated, further research and development remains before the technology is widely available (Carter 2013).

4.4 Relative Costs of Desalination Technologies

Among the choice of different technologies, RO is today's state-of-the-art technology for high-quality water. RO is the most developed as well as the cheapest way to convert brackish water or seawater into drinkable water, as we will now show.

The cost of a desalting plant is determined by a number of technical and economic factors. The major cost categories are capital costs and operating and maintenance costs. Capital costs are determined by the process type, plant capacity, feed-water type and salinity, pretreatment required, product salinity desired, and site-related costs for land, plant, and brine disposal. Operating and maintenance costs include labor, energy, supplies, and general administrative expenses. The economic characteristics of a desalting plant are usually expressed in two ways: the capital cost per unit of installed capacity, such as dollars per gallon per day; and the total annual product water costs, such as dollars per thousand gallons of annual production. The product water costs are determined by the ratio of the total annual costs to the annual water production. Since annual costs (including fixed costs) are incurred at some minimum level even when no water is produced, the product water cost is sensitive to the level of output and can increase significantly if the output drops. High water costs often result from excessive "down time" for maintenance, occasionally from shortages in suitable feed water, and from fluctuation in product demand, all of which decrease annual output.

Dual-purpose plants can affect the economics of desalination. For example, in a dual-purpose electric power/desalination plant, waste heat from electric power production can be used for distilling seawater in the desalting plant, or steam pressure from power production for desalination by RO.

Water costs of existing desalination plants vary widely, and frequently are not completely comparable because of differences in the cost-determining factors mentioned above. Nevertheless, the three factors that are comparable and have the largest effect on the cost of desalination per unit of fresh water produced are the feed-water salinity level, energy costs, and plant size, which show economies of scale (Winter et al. 2002).

4.4.1 Feed-Water Salinity Level

An increase in the salt content of the feed-water increases the operating costs as desalination takes longer and/or uses more equipment. Brackish water can be desalted most economically on a large scale by RO (Larson and Leitner 1979; Glueckstern and Kantor 1983). RO brackish water desalination costs about \$1.50–\$2.50 US per 1,000 gallons. As shown by Dore (2005), seawater desalination on a large scale by RO can be obtained for about US \$0.5 per cubic meter.

4.4.2 Energy Requirements

The energy required for desalination can represent about 44 percent of operating costs under distillation as described above (see Fig. 4.1). The energy requirements for employing various RO and Distillation treatments are presented in Table 4.1 and clearly show that RO requires the lowest energy demand. This is a result of immense heating required for water evaporation in the process of distillation. For this reason, between the two best technological options (i.e. RO and distillation), RO takes the lead, also in seawater desalination (Sackinger 1982 and Glueckstern 1999). Wood (1982) observed that rising world energy prices would alter the relative costs of different desalination methods, increasingly favoring RO. But the development and application of renewable energy could change this picture. For example, in Australia, the Perth Desalination Plant produces up to 38 million gallons per day using power from the Emu Downs Wind Farm. The 48 turbines at the wind farm produce 80 MW/day, more than three times the needs of the plant (Mydens 2007). Since 2008, Australia has added five large-scale desalination plants. These facilities have a production capacity ranging from 36 to 120 million gallons per day and all use renewable energy sources including wind, solar, and wave energy (Furukawa 2013).

4.4.3 Economies of Scale

According to Mielke, the economies of scale favoring larger plants are particularly significant for plants with capacities smaller than 3 million gallons per day (Mielke 1999). The distillation processes benefit most from economies of scale, while for RO such economies of scale lead to a fall in unit costs at a lower rate. Nevertheless,

Table 4.4 RO desalination plants and their costs

#	Plant name, location, and source	Plant size (MGD)	Average total cost ($US per 1,000 gallons)	Total cost per year (in millions of $US)
1	Plant #1, USA, using seawater[a]	0.01	13.42	0.05
2	Plant #2, USA, using seawater[a]	0.1	9.88	0.36
3	Plant #3, USA, using seawater[a]	1	7.40	2.70
4	Plant #4, USA, using seawater[a]	1	1.67	0.61
5	Plant #5, USA, using seawater[a]	3	6.64	7.27
6	Plant #6, USA, using seawater[a]	3	1.41	1.54
7	Plant #7, USA, using seawater[a]	5	1.33	2.43
8	Plant #8, USA, using seawater[a]	5	6.36	11.61
9	Plant #9, USA, using seawater[a]	10	1.23	4.49
10	Plant #10, USA, using seawater[a]	10	6.03	22.01
11	Plant #11, USA, using seawater[a]	25	1.21	11.04
12	Plant #12, USA, using seawater[a]	25	5.96	54.39
13	Desalination unit at the Chevron Gaviota oil and gas processing plant (seawater, USA)[b]	0.45	16	2.63
14	Desalination plant in the City of Morro Bay (brackish water, USA)[b]	1.20	7 (Avg = 1,750/AF)	3.07
15	Desalination plant in the City of Santa Barbara, Goleta, and Montecito (seawater, USA)[b]	5 (7,500 AF/year)	7.67 (Avg = 1,918/AF)	14.00
16	Desalination facility in the Monterey Bay Aquarium (seawater, USA)[b]	0.04	5.53 (1,800/AF)	0.09
17	Desalination facility on the Santa Catalina Island (brackish water, USA)[b]	0.13	6 (2,000/AF)	0.29
18	Sand city (seawater, USA)[c, i]	3	N/A	2.18
		6	N/A	3.82
19	MRWPCA Santa Cruz Moss landing sites (seawater, USA)[c]	7	N/A	5.35
		14	N/A	9.56
20	RO desalination plant in Florida (seawater, USA)	5 (25,000 m^3)	N/A	46.53
21	RO desalination plant in Florida (brackish water, USA)	5 (25,000 m^3)	N/A	11.79
22	Design center models (brackish water, USA)[d]	0.26	2.3	0.22
		1.06	1.5	0.58
23	Lower Valley Grand Cayman (seawater, Caribbean Islands)[e]	0.4	7.16	1.05
24	Aqua design ltd. (seawater, British Virgin Islands—Tartola)[f]	1	16.50	6.02
		1	15.80	5.77

(continued)

Table 4.4 (continued)

#	Plant name, location, and source	Plant size (MGD)	Average total cost ($US per 1,000 gallons)	Total cost per year (in millions of $US)
25	Aqua design ltd. (Brackish water, British Virgin Islands—Tartola)[f]	1	9.10	3.32
26	Aqua design ltd. (seawater, British Virgin Islands—Virgin Gorda)[f]	0.02	13.10	0.10
27	Tampa Bay Florida (seawater, USA)[g]	25 (94,625 m^3/d)	2.45	22.36
28	Eilat (seawater, Israel)[h]	2.6	2.72	2.58
29	Eilat, second plant (seawater, Israel)[h]	2.6	3.06	2.90
30	Larnaca (seawater, Cyprus)[h]	10.6	3.14	12.15
31	Pasadena—California desalination plant (Seawater, USA)[i]	70	4 (1,000/AF)	102.20
32	The Aqaba hybrid scheme (seawater, Jordan)[j]	64.5	3.45	81.22

Sources
[a] US Congress, Office of Technology Assessment (1988)
[b] California Coastal Commission (1993)
[c] Parsons Engineering Science and American Water Works Service Company (1997)
[d] Grethe and Beltle (1993)
[e] Andrews et al. (1998)
[f] Government of the British Virgin Islands (1995)
[g] Leitner (1999)
[h] Wilf and Klinko (1995)
[i] Parsons Engineering Science (1996)
[j] Glueckstern (1982)

the economies of scale can reduce unit costs by as much as 55 percent when using RO to treat seawater, according to US Congress, Office of Technology Assessment that conducted a thorough investigation of desalination technologies in 1988.

In order to determine if economies of scale are present, we used the information on costs from 36 plants that use RO for their desalination operations. Table 4.4 presents the dataset obtained. For each plant, the table shows the plant number or name, its location, and whether it desalinates brackish or seawater; plant size in millions of gallons per day (MGD) of freshwater output; average total costs in $US per 1,000 gallons; and the total cost per year (in millions of $US).

Finally, to determine whether economies of scale exist for RO desalination, the average total cost (ATC) can be plotted with respect to plant size. Dore (2005) uses 12 data points when verifying the presence of economies of scale in RO. Figure 4.12, on the other hand, makes use of an expanded data sample, composed of the 36 observations presented in Table 4.3. Figure 4.12 also distinguishes between brackish water and seawater RO desalination.

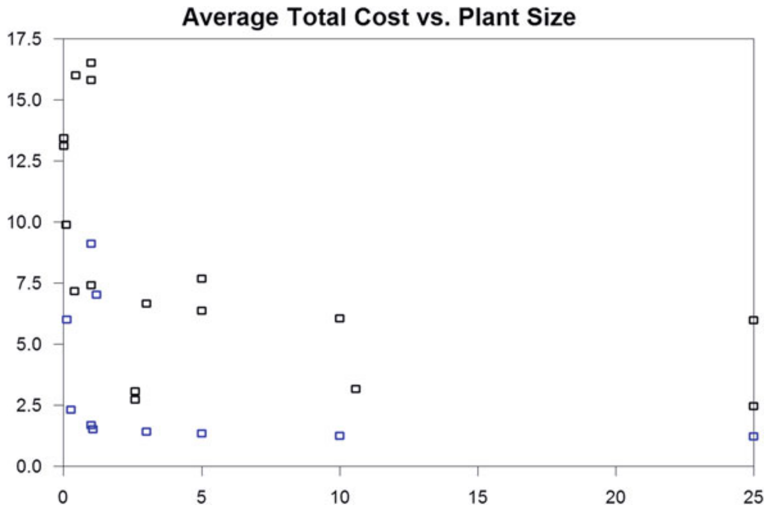

Fig. 4.12 Reverse osmosis economies of scale for brackish and seawater

In Fig. 4.12, seawater desalination ATC are represented by black inscription and brackish water desalination ATC by blue inscription. This figure clearly shows that in the case of both brackish and seawater desalination, an increase in treatment capacity results in a per unit decrease in cost.

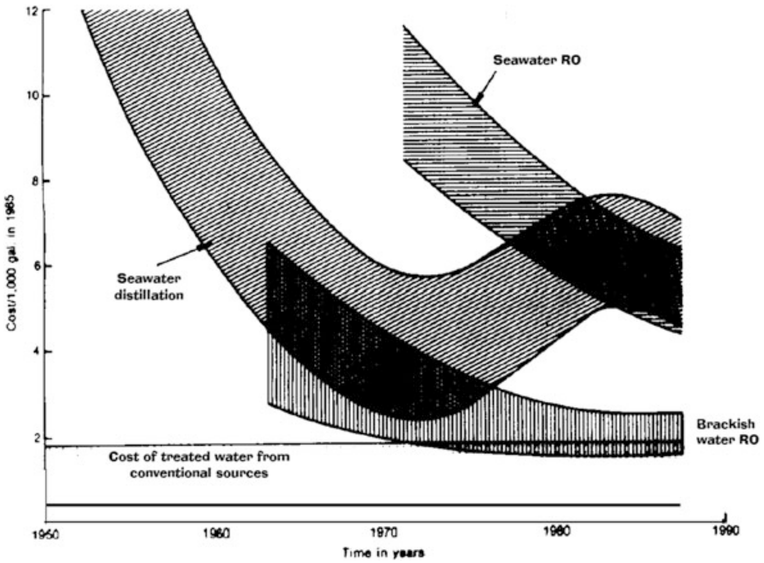

Fig. 4.13 Real desalination costs over time (US Congress, Office of Technology Assessment 1988)

To summarize the cost of RO technology relative to the costs of other dominant technologies used in freshwater production, we use Fig. 4.13. This figure illustrates the steadily decreasing real costs of RO desalination, as opposed to the volatile changes in distillation costs, and the increasing costs of conventional water treatment (non-desalination methods) from 1940 to 1985. It is shown that RO costs have been rapidly falling over the past decades. In addition, brackish RO desalination costs fell below the distillation costs in the 1970s, and could undercut the costs of conventional water treatment.

4.5 Conclusion

The early US government support for R&D in desalination together with easy access to the technology contributed to the long-lasting decrease in costs of desalination. Due to the development of membrane technology and finally RO in the early 1980s, desalination costs have fallen significantly and the forecast is a continued fall in costs over time in the near future. Based on this forecast, and the fact that the costs of obtaining and treating water from conventional sources have been increasing, we can expect that desalination costs will become competitive with those of conventional water treatment processes and that RO and other membrane methods will become competitive for freshwater production.

The technology that is the most economical of all the desalination methods is RO technology (Mesa et al. 1996). Compared to other technologies, only distillation can rival RO in seawater desalination, due to superior economies of scale and much lower up-front investment costs. This, coupled with distillation's maturity and reputation for reliability (Winter et al. 2002), gave an early lead to distillation plants. However, the new developments in RO manufacturing have now made it the dominant method worldwide. RO plants are already replacing distillation plants all over the world (Winter et al. 2002). RO has several advantages over other desalination technologies including lower energy requirements, fewer problems with corrosion, higher recovery rates for seawater, and less surface area for the same amount of water production (Abulnour et al. 1983). The ability to produce potable drinking water for significantly less than \$1.00 per m^3 (Mielke 1999) is by far its greatest asset. In Singapore, the seawater RO plant, with a capacity to produce 136,000 m^3/day, in operation since 2005, under a public-private scheme, is producing desalted water at 0.78 US \$/m^3.

Desalination is shown to be the increasingly prefered method of freshwater production and, especially as technical advancements of membrane processes improve their costs and efficiency, RO will continue to be the prefered choice for countries moving into desalination (Winter et al. 2002). For this reason we have chosen RO technology as the basis for the analysis of pricing and econometric estimation of Ramsey Pricing, which follows in Chap. 5.

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Chapter 5
The Theory of Water and Utility Pricing

5.1 Introduction

In this chapter, we survey the theory and practice of the pricing of water as a public utility. Section 5.2 reviews the classic theory of marginal cost pricing developed by Dupuit (1854) and expanded by Hotelling (1938). Marginal cost pricing maximizes consumer welfare and to this day, even after major developments in public economics from the 1970s to the present, survives as the most enduring guiding principle of utility pricing. Marginal cost pricing requires that the capital costs be covered by general taxation. In Sects. 5.3, 5.4 and 5.5, we consider pricing where the utility cannot rely on the capital costs being covered by the state or by a higher jurisdiction; this is Ramsey pricing, an idea that was first put forward in Ramsey (1927). In the climate of low taxation, and a declining role of the state, many higher level jurisdictions now do not offer full coverage of capital costs of the drinking water treatment plants run by municipalities, and so the latter are obliged to charge prices higher than marginal costs to cover the capital costs. We consider utility pricing *in the context of* recent developments in the "new" public economics that can be dated as beginning with the work of Mirrlees (1971), and show that while Ramsey pricing remains relevant to the modern utility, it can be *integrated* with the new public economics by introducing "shadow Ramsey Pricing," where state subsidies can range from positive to zero for capital costs. We restate the reformulated theory of Ramsey pricing and illustrate diagrammatically the incorporation of shadow prices. In Sect. 5.6, Ramsey prices and Shadow Ramsey Pricing are computed using data from reverse osmosis treatment plants presented in Chap. 3. Thus this section demonstrates that utility pricing that is compatible with the new public economics is possible in practice. In Sect. 5.7, we consider actual pricing practice in some developed OECD countries, such as the USA, European Union, and Australia, and assess how far the practice deviates from the theoretical principles of utility pricing. Section 5.8 brings together the main lessons for the pricing of water.

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5.2 The Dupuit-Hotelling Theory of Marginal Cost Pricing

In economic theory, the standard approach to pricing in public utilities is that price should equal marginal costs, as this approach maximizes consumer surplus. This policy prescription follows the seminal contributions by Dupuit (1854) and Hotelling (1938). To quote from Hotelling (1938):

> The common assumption, so often accepted uncritically as a basis of arguments on important public questions, that "every tub must stand on its own bottom," and that therefore the products of every industry must be sold at prices so high as to cover not only marginal costs but also all the fixed costs, including interest on irrevocable and often hypothetical investments…[This is] inconsistent with the maximum of social efficiency.

He applied this principle to "all bridges, roads, railroads, waterworks, electric power plants, and like facilities"; in other words, facilities supplied by the public sector, irrespective of the level of jurisdiction. Thus on social efficiency grounds, Hotelling, following Dupuit (1854), argued that the capital costs must be met through general state revenues such as income and inheritance taxes, and land taxes. What follows is an exposition of the marginal cost pricing rule as given by Hotelling (1938).

5.2.1 The Derivation of the Marginal Cost Pricing Rule

Let there be n commodities q_i and n prices p_i. Then the demand functions are:

$$p_i = f_i(q_1, q_2, \ldots, q_n), \quad (i = 1, 2, \ldots, n) \tag{5.1}$$

Then the consumers' surplus is

$$\int (f_1 \mathrm{d}q_1 + f_2 \mathrm{d}q_2 + \cdots + f_n \mathrm{d}q_n) \tag{5.2}$$

taken from an arbitrary set of values of the q's. The net benefit is obtained by subtracting from (5.2) a similar line integral of the marginal cost functions:

$$g_i(q_1, q_2, \ldots, q_n)$$

Now let

$$h_i = f_i - g_i$$

Then the total net benefit is:

$$w = \int h_i \mathrm{d}q_i \tag{5.3}$$

Let the function Φ represent the n-good hyper-indifference surface, along which utility is held constant. Note that Φ can be replaced by any increasing function Ψ of Φ.

Let the individual budget constraint, after taxes, be

$$\sum p_i q_i = m, \ \text{(where } m \text{ is money income)} \tag{5.4}$$

Assuming utility maximization, the consumer chooses the highest Φ, subject to (5.4).

Suppose $q_1^{'},\ q_2^{'},\ \ldots, q_n^{'}$ were any other set of quantities satisfying (5.4), so that:

$$\sum p_i q_i^{'} = m \tag{5.5}$$

Then:

$$\Phi = \Phi(q_1, \ldots, q_n) > \Phi\left(q_1^{'}, \ldots, q_n^{'}\right) = \Phi + \delta\Phi$$

Let us consider a system with the imposition of excise taxes and *reduction* of income taxes. Of course some of the taxes may be negative. The excise tax may be called a "toll" or a "service user fee" *over and above* marginal cost. Now there may be a redistribution of production and consumption. Let p_i, q_i and m be replaced, respectively, by

$$p_i^{'} = p_i + \delta p_i, \quad q_i^{'} = q_i + \delta q_i, \quad m^{'} = m + \delta m \tag{5.6}$$

where the increments can be either positive or negative. The new excise tax revenue is $\sum q_i^{'} \delta p_i$. The consumer's income tax is reduced by δm, and the net increment of government revenue is

$$\delta r = \sum q_i^{'} \delta p_i - \delta m \tag{5.7}$$

The consumer's budget constraint is $\sum p_i^{'} q_i^{'} = m^{'}$, which we can also write as

$$\sum (p_i + \delta p_i)(q_i + \delta q_i) = m + \delta m \tag{5.8}$$

Subtract the budget Eq. (5.4) corresponding to the former regime (excise taxes) and using (5.6) we find that

$$\delta m = \sum q_i^{'} \delta p_i + \sum p_i \delta q_i \tag{5.9}$$

Next substitute (5.9) into (5.7) and we get

$$\delta r = -\sum p_i \delta q_i \tag{5.10}$$

Now using the definitions in (5.6) (i.e. put $q_i^{'} = q_i + \delta q_i$) into (5.5) and subtract (5.4). Then for any set of values of $\delta q_1 \ldots\ \delta q_n$, we have:

$$\sum p_i \delta q_i = 0 \tag{5.11}$$

Then it follows that:

$$\delta \Phi = \Phi(q_1 + \delta q_1, \ldots, q_n + \delta q_n) - \Phi(q_1, \ldots, q_n) < 0 \tag{5.12}$$

Suppose that the excise tax paid is exactly offset by the reduction of this income tax. Then $\delta r = 0$. From (5.10), it follows that (5.11) is satisfied.

Except in the highly improbable case of all the δq's coming out exactly zero, it would follow from (5.12) that the consumer is on a "lower" indifference surface than before. The change from income to excise taxes has resulted in a net loss of satisfactions. QED.

Hotelling concludes at the end of his theorem (retaining his italics):

> If government revenue is produced by any system of excise taxes, there exists a possible distribution of personal levies among the individuals of the community such that the abolition of the excise taxes and their replacement by these levies will yield the same revenue while leaving each person in a state more satisfactory to himself than before.

He then proceeds to measure the welfare loss as being approximately equal to:

$$\Delta w = 1/2 \sum \delta p_i q_i$$

Hotelling also presciently predicts that two groups of people are likely to object to this social policy: (5.1) the very rich, who normally would pay more taxes than the poor, and (b) "land speculators" (Hotelling 1938, p. 259). For the rich, it could be the case that the personal benefit to them of the marginal cost principle may be lower than when there is a toll or excise tax. Land speculators, whose land is adjacent to the bridge, would prefer to collect the toll rather than have the bridge paid for out of general taxes. When the political climate moves away from considerations of "general human welfare" and wellbeing to a climate of "low taxes," then it is clear what the dominant voices are in shaping public policy: it is the rich and the sectional interests who stand to gain by a policy of tolls and service charges for the outputs of public utilities.

This principle of marginal cost pricing has been the bedrock of public utility pricing in most textbooks that cover the public sector. However, in most developed economies, led by the example of the United States, there is pressure to reduce all taxes, especially income taxes, capital gains taxes, and corporate taxes, and replace the revenue from either "user fees," or sales and consumption taxes. Some of this impetus has come from economists who emphasize that taxing income is "distortionary" and also not "incentive-compatible" and that not taxing income would encourage "saving," on the assumption that all savings will be invested, for the betterment of humanity.

But this principle of marginal cost pricing has increasingly become of theoretical interest only. With declining tax revenues and also budget deficits, the principle is being abandoned in most jurisdictions with a few exceptions. Almost all local jurisdictions are told to raise their own capital or charge for it through user fees, i.e. they are told that they are "on their own." For example, in Ontario (see Dore 2015), all municipalities are legally required to plan for the full cost of their water including the capital costs and also for the future renewal of water infrastructure. In Europe and elsewhere there is even pressure to "privatize" water utilities, as many of the state-controlled (in the case of water, mostly municipal) utilities do not have the finance to renew their infrastructure and have even neglected infrastructure maintenance, called "deferred maintenance." Much of North America has this problem of old and failing water infrastructure. The political climate of low taxes and privatization is a pervasive phenomenon, covering most developed countries, i.e. countries in the European Union, the US as well as Canada.

Is there a case for the privatization of the drinking water sector? The next section considers the theoretical case for privatizing what has been previously public sector production. The main case against public provision is the possible inefficiency of the public sector, which does not have profitability as a guide to all decision making.

5.3 Private Versus Public Production

The choice between private or public production should be based within the general framework of what is now called the "new public economics" (Boadway 1997) when Professor Jim Mirlees opened up a new avenue of research with the theory of optimal taxation (Mirrlees 1971). The advocates of optimal taxation draw their inspiration from two papers by Ramsey (1927, 1928). In his 1928 paper, Ramsey was trying to work out how much a "socialist society" would need to save, a society that would have a "planning authority," with a well defined *single social welfare criterion that it wished to maximize.* But what might have been appropriate for a single-minded socialist planning authority has now been transformed into an inquiry in which a heterogeneous society is analyzed by treating it as a *single* consumer. That is, in this approach, social questions are answered by positing a single "representative agent." This is a microeconomic approach, using a representative consumer, or assuming that all consumers are alike; in Ramsey, this was an interesting thought experiment. This chapter has given rise to the theory of "optimal economic growth." However, a year earlier (Ramsey 1927) published an article on commodity taxation and pricing, when a utility cannot implement the Hotelling marginal cost pricing rule but must raise prices *above* marginal cost in order to cover the capital costs. The pricing rule that emerges from Ramsey takes into account the need to raise the necessary capital, and so "constrain" the optimal prices, called Ramsey Prices, for public utilities. The pricing rule can be called the "optimal departure" from marginal cost pricing, where the higher prices are charged on those whose elasticity is the lowest, whether it is the demand elasticity or the supply elasticity. Ramsey did not address the question as to whether the production was to be produced in the public sector or the private sector. That question became important in an era with a drive toward deregulation and a major effort to make government "small." An understanding of the historical context might make it easier to understand this new direction whereby the Central State reneges on its responsibilities and requires lower local jurisdictions to assume the financial burdens which were previously shouldered by the Central Government.

The low-tax climate referred to above was largely responsible for the diminution of the Central State, an ideological position championed by President Reagan in the US and Margaret Thatcher in the UK. For example, the Conservative government of Margaret Thatcher in the UK undertook large-scale privatization; and the sale of state assets saw the reduction of the share of public production of goods and services in GDP from 6.6 percent in 1982 to 1.9 percent in 1991, compared to a share of 0.6 percent in the United States in 1987. The benefits of privatization were thought to be increased efficiency and benefits for consumers. A survey of the evidence carried out by Boardman and Vining (1989) supported this view. In economic theory, the relative role of the public and private sectors remained a key issue in public finance and "public economics." Some fundamental theoretical questions on this have been well addressed in a balanced manner by Stiglitz (1989). Nevertheless in the economics of the Chicago school, there appears to be a strong bias in favor of privatization. Although John Maynard Keynes (1981) was the main proponent of the role of the state in economic affairs, he was not dogmatic on the boundary between the private and public sectors. To quote Keynes (1981):

> The line of demarcation between the two is constantly changing in accordance with the practical needs of the day. As to where precisely this line should be drawn no great question of principle is involved.

The balanced approach of Stiglitz indicated that the relative attractiveness of public ownership and production depended on the importance of market failures versus public failures. Failures in either sector can arise on account of a lack of competition, imperfect information, and incomplete markets. Taking this literature into account, several papers formalized the choice between private and public production. Shapiro and Willig (1990), Bos and Peters (1988, 1991), and Hart et al. (1997) focus on the market failures of imperfect information and incomplete contracts; Laffont and Tirole (1991) and Schmidt (1996), in addition, consider the public failure of governments that fail to live up to their pre-announced commitments.

While these papers are insightful, they do not offer an explanation of why the share of public production varies so much even among developed countries. The theory of optimal taxation offered a fruitful approach to the question (see, for example, Atkinson and Stern (1974), Dasgupta and Stiglitz (1972) Diamond and

Mirrlees (1971a, b). A key idea in the demarcation of private or public production is that the optimal choice between public and private be determined by "absolute efficiency advantage," (Huizinga and Nielsen 2001). However, as Huizinga and Nielsen's paper is concerned with the privatization decision, they do not investigate the characteristics of *absolute efficiency advantage*; in their paper, the line of demarcation between public and private production depends on a "waste" parameter, within a framework of optimal taxation. Therefore it would be interesting to develop the necessary conditions for absolute efficiency advantage, conditions that would be consistent with Huizinga and Nielsen, and with the theory of optimal taxation. Dore et al. (2004). examined the question of whether the "public utility" production should be carried out in a public enterprise or whether it should be carried out by the private sector. Accordingly, the Ramsey-inspired theory suggests that the decision should be based on which form of enterprise has *absolute efficiency advantage.* The section below draws on Dore et al. (2004).

5.4 Absolute Efficiency Advantage

First, an enterprise that had *absolute efficiency advantage* would be free of failures of both market and public types, and hence there would be no negative externalities. Under competitive conditions, absolute efficiency advantage would yield a higher consumer surplus. (Otherwise it would contradict the definition of efficiency). The assumption of perfect competition means that the technology used would be the "state of the art" technology, assuring the optimal quality of the product. Furthermore, under perfect competition, there are no externalities, by assumption. That is, the lower cost and/or quality are not obtained at the expense of some other social cost, such as the degradation of the environment or the shifting of social costs to a future generation. These considerations suggest the following necessary conditions for absolute efficiency advantage: an enterprise would have absolute efficiency advantage if: (a) its product is superior in terms of quality, (b) it can supply the good at a lower unit price, and (c) the production does not entail any negative externalities. If these three conditions were met, then the consumer surplus would be the highest possible. The consumer surplus cannot be higher unless the waste of resources is at a minimum. Indeed these necessary conditions would also be necessary and sufficient under conditions of perfect competition. (Of course at the social optimum, characterized by competitive general equilibrium, waste would be zero, and all potential gains exhausted, with all resources allocated to their highest marginal values.)

However, in a second-best (Ramsey) world, it would be safer to state the same three conditions as *necessary* conditions, not sufficient conditions. In a second-best world, there are distortionary taxes paid by the private sector, monopolistic competition and market power, unionized labor, etc. Both product quality and unit cost would also depend on the technology used, and the quality of management may vary. To repeat, economic theory recognizes that taxation has distorting effects, and of course private enterprise would be subject to taxation. Thus when a product is produced in the private sector, the absolute efficiency advantage should remain even after taxation, since all agents are taxed. Hence, for goods with some characteristics of a public good (such as drinking water), there may be a presumption that the distortionary effects of taxation can be avoided if the good is produced in the *public* sector, assuming of course that there is no public failure. On the other hand, if the product is produced in the private sector, then the absolute efficiency advantage must be demonstrated.

The demonstration must focus on the necessary conditions. The third of the three necessary conditions (no negative externalities) would in general be hard to demonstrate conclusively, but some evidence may nevertheless be available. In the privatization of drinking water production, two countries are often cited: the UK and France. Dore et al. (2004) reexamined the evidence of the consequences of privatization in terms of absolute efficiency advantage in both countries. They used the necessary conditions for absolute efficiency advantage stated above. They showed that in water production, absolute efficiency advantage requires that we consider product quality, unit price, and whether there is evidence of negative externalities. The theory indicates that each of the three necessary conditions must be met in order to demonstrate absolute efficiency advantage. Dore et al. (2004) next investigated the impacts of privatization.

In the UK, privatization resulted in significant environmental improvements. The massive investments by the 10 regional water firms improved the quality of drinking water and the country's waterways and increased the number of beaches at which it was safe to swim. Compliance with European standards also improved from 76 percent in 1989 to almost 92 percent in 2000. These improvements, however, were not without cost as water bills also increased substantially. The question arises as to what increases would have happened if the utilities had not been privatized. A definitive answer is virtually impossible. We do know that the price increases exceeded the rate of inflation, but once again, we do not know what the increase would have been under a publicly owned system. We can, however, compare the privatized utilities in the UK with publicly owned utilities elsewhere. For example, the average rate of return for the English companies was approximately 3 times that of publicly owned Swedish companies. Over the period, the total pre-tax profits for the 10 UK water companies increased on average by 142 percent (Dore et al. 2004). Comparing these profit margins with those in Sweden, Spain, Hungary, and France indicates much higher profits for the English firms. Suffice to say, the profit rates in the UK were extremely high by international standards. It is therefore not surprising that the regulatory bodies eventually found the increases to be unwarranted and ordered rebates, and the abnormal profits were taxed away. We may conclude that although the environment improved, in the case of the UK the record does not show that the private sector had any decisive absolute efficiency advantage.

Unlike the UK privatization, the water and sewage systems in France are publicly owned but in large part privately operated by a variety of contracts. As in the UK, the main policy objective was compliance with the new European Union standards. To achieve this objective, a polluter-pay strategy was adopted, albeit with inherent problems. The tax was not related to the amount of effluent discharged and the fines were returned to the polluters. The overall enforcement and monitoring were uncoordinated and inadequate because the regulatory functions were diffused among different agencies and the 36,500 geographic levels of jurisdictions. In addition, management contracts between local communes and private companies were not monitored by any agency with the appropriate technical and economic resources. For example, contracts were not properly tendered and costs were inflated to justify higher prices. Moreover, the privately run companies charged higher prices for water than the publicly managed utilities, on average 40 percent higher. Thus it seems clear that the French water model lacked the proper machinery for economic regulation.

To conclude, although water quality improvements were associated with privatization, there is no demonstrable evidence that privatization resulted in lower prices. In fact, the evidence in both countries indicates *higher* prices because of privatization. It should be noted that the experience in both countries is similar to the privatization of local hydro utilities in the Province of Ontario, Canada, where costs increased significantly due to a similar private sector tenet of maximizing shareholder value. It seems that the regulatory system in England and France did not work satisfactorily for many years after privatization. With natural monopolies in water, private production requires adequate regulation. In the two countries examined, it is not possible to find that the private sector demonstrated absolute efficiency advantage.

Nevertheless it should be noted that new water technology has always been developed by the private sector and not by the public sector, which typically is not allowed to do any innovation or R&D. The new technology is developed by a *manufacturing sector* not connected with the provision of water. With efficient markets, the private sector that develops new technologies should have been able to sell these technologies to the public sector. Once the central state had reneged on its responsibilities for financing water infrastructure, the main local government failure of the public sector (before privatization) was the failure to anticipate the revenue requirements that would *no longer be forthcoming from the central state,* and that therefore they would have to charge prices for water services in order to plan and budget for investments in new infrastructure. Under these conditions, it is not at all clear that in the UK and France the only solution to this problem was privatization.

Nevertheless it is possible that private sector *management contracts* could manage some utilities better by implementing cost control and reducing waste and duplication, or reducing redundant labor. Thus it is the quality of management that counts, not whether the utility is in the public sector or in the private sector.

5.5 Second-Best (Ramsey) Pricing

In economic theory, Marginal Cost Pricing is identified as the "first-best" solution that maximizes welfare. However, given the hierarchical jurisdiction of public utilities, and the requirement of self-financing, this first-best solution may not be possible if the capital cost is not provided by the central government. In such a situation, it is appropriate to consider the "second-best" Ramsey Pricing.

Briefly, Ramsey prices are prices that are Pareto optimal subject to a constraint on the total profits of a single supplier or group of suppliers. In particular, because a utility whose activities are characterized by scale economies will lose money if it sets the prices of its products equal to their marginal costs, Ramsey prices become for that utility the prices that are optimal (economically efficient) given the financial feasibility requirement that the firm's profits be non-negative. The same Ramsey prices can also be shown to be those necessary for maximization of the sum of consumers' and producers' surpluses.

5.5.1 Derivation of Ramsey Prices

The exposition of Ramsey pricing given here follows that of Baumoland Bradford (1970).

As before let $x_1, \ldots, x_n$ be the quantities of n goods produced by a natural monopoly and let $p_1, \ldots, p_n$ be the corresponding prices. Let $Z(p_1, \ldots, p_n)$ be the consumer's indirect utility function. The natural monopolist now has a profit constraint:

$$\Pi(p_1, \ldots, p_n) = M \tag{5.13}$$

Maximize $Z(\cdot)$ subject to (5.13):

$$\text{Max } Z(p_1, \ldots, p_n) + \lambda[M - \Pi(p_1, \ldots, p_n)]$$

For a maximum,

$$\frac{\partial Z}{\partial p_i} = \lambda \frac{\partial \Pi}{\partial p_i}, \quad i = 1, 2, \ldots, n \tag{5.14}$$

Equation (5.14) says that the marginal gain from a given price change must be proportionate to the marginal profit cost. Equivalently, for all goods produced, the ratio of marginal gain to marginal profit cost must be the same.

From consumer demand theory, we can also show that

$$\frac{\partial Z}{\partial p_i} = -x_i \tag{5.15}$$

Substitute into (5.14) to get

$$-x_i = \lambda \frac{\partial \Pi}{\partial p_i} \tag{5.16}$$

or

$$\frac{1}{\lambda} = -\frac{1}{x_i}\frac{\partial \Pi}{\partial p_i}$$

For any other good j, we have

$$\frac{1}{\lambda} = -\frac{1}{x_j}\frac{\partial \Pi}{\partial p_j}$$

Equating them, we get

$$\frac{1}{x_i}\frac{\partial \Pi}{\partial p_i} = \frac{1}{x_j}\frac{\partial \Pi}{\partial p_j} \tag{5.17}$$

Equation (5.17) says that the ratio of marginal profit is the same as the ratio of the two output quantities. This is a Ramsey result.

The same result can be expressed as a price deviating from marginal cost (MC). Let marginal revenue for product i be MR_i, and let E_i be the price elasticity of demand for good i.

Note that MR_i is:

$$\mathrm{MR}_i = p_i + x_i\frac{\partial p_i}{\partial x_i} \tag{5.18}$$

and

$$\begin{aligned}\frac{\partial \Pi}{\partial p_i} &= (\mathrm{MR}_i - \mathrm{MC}_i)\frac{\mathrm{d}x_i}{\mathrm{d}p_i} \\ &= \left(p_i + x_i\frac{\partial p_i}{\partial x_i} - \mathrm{MC}_i\right)\frac{\mathrm{d}x_i}{\mathrm{d}p_i}\end{aligned} \tag{5.19}$$

Substitute (5.19) into (5.16):

$$-x_i\frac{\mathrm{d}p_i}{\mathrm{d}x_i} = \lambda\left(p_i + x_i\frac{\mathrm{d}p_i}{\mathrm{d}x_i} - \mathrm{MC}_i\right)$$

Now adding $(p_i + x_i\frac{\mathrm{d}p_i}{\mathrm{d}x_i} - \mathrm{MC}_i)$ to both sides of the above equation, we get

$$\begin{aligned}p_i - \mathrm{MC}_i &= (1+\lambda)\left(p_i + x_i\frac{\mathrm{d}p_i}{\mathrm{d}x_i} - \mathrm{MC}_i\right) \\ p_i - \mathrm{MC}_i &= (1+\lambda)(\mathrm{MR}_i - \mathrm{MC}_i)\end{aligned} \tag{5.20}$$

Equation (5.20) says that the difference between price and marginal cost should be proportionate to the difference between marginal revenue and marginal cost.

We can rewrite (5.20) as

$$-\lambda(p_i - \mathrm{MC}_i) = (1+\lambda)x_i \frac{\mathrm{d}p_i}{\mathrm{d}x_i}$$

or

$$\frac{(p_i - \mathrm{MC}_i)}{p_i} = \frac{(1+\lambda)}{\lambda}\frac{1}{E_I} \tag{5.21}$$

Since

$$E_i = -\frac{x_i}{p_i}\cdot\frac{\mathrm{d}p_i}{\mathrm{d}x_i}$$

Equation (5.21) says that the "markup" over marginal costs should be proportionate to the inverse of the elasticity; the more *inelastic* the demand, the higher the markup.

While some considered Ramsey pricing to have been a path-breaking contribution to economics, its principles were largely forgotten even though it was rediscovered and expanded upon by Pigou, Boiteux, and Samuelson. Its history was explored in an article by Baumol and Bradford (1970), and the principle has since been widely recognized and accepted by economists and practitioners. For example, in 1983 the US Interstate Commerce Commission adopted Ramsey pricing as the underlying principle it would follow in the regulation of railroad rates. The American Water Works Association (AWWA) also seems to favor Ramsey Pricing, although they do not call it that (Overcast 2012).

> To quote Baumol and Bradford (1970):
> Ramsey prices are an outstanding example of the use of pure economic theory to derive an operational solution to a difficult set of practical problems. It may also be as definitive as any second-best theorem (Baumol and Bradford 1970, p. 88).

Pareto optimality requires that the prices be those which elicit such a set of outputs and purchase quantities that it is impossible to increase the welfare of any one individual without harming anyone else. The definitive character of Ramsey pricing is surprising in the light of the conclusions suggested by much of the "second-best" literature, that where additional constraints are superimposed on the usual requirements of optimality, one can expect no simple and straightforward results to emerge (Lipsey and Lancaster 1956). The Lipsey-Lancaster result is important; it states, roughly, that when there are many factors that make the economy depart from optimality, "fixing" any one will not get the economy "closer" to the social optimum. Thus "piecemeal" optimization cannot bring welfare to being "closer" to the first-best optimum which is the competitive general equilibrium.

The question of what is "first-best" and how and whether it can be approached in a piecemeal manner to arrive at a second-best optimum has been to a large measure part of the research agenda of the New Public Economics, which started with the publication of the work of Mirrlees (1971). Much of the new research in public economics, inspired by Ramsey's idea of a "second-best solution" that is Pareto-wise improving, shows that policy makers and state tax authorities face asymmetric informational constraints which means that the "optimal" redistribution envisaged by the Second Theorem of Welfare Economics is just not possible. This is partly because the framers of the Second Theorem, in their zeal for justifying a decentralized free enterprise economy, ignored the fact that the Second Theorem was incentive-incompatible, and therefore essentially vacuous. This has now led researchers like Guesnerie (1994) and Boadway (1997) to conclude that society's second-best efficiency frontier lies *everywhere* inside the first-best frontier. This result is important enough to be called a new "theorem of new public economics" that should replace the Second Theorem of Welfare Economics; I shall call it the *Third Theorem of (Public) Welfare Economics,* a theorem that is the logical result of the whole new public economics originating with the work of Professor James Mirrlees.

Given the Third Theorem, most policy rules applicable to first-best economies no longer apply for real economies: social values (or shadow prices) differ from market prices; the standard first-best Samuelson rules (for public expenditures) are generally no longer valid; and quantity controls may be efficient policy instruments. But there is more bad news: in a dynamic world, even this second-best frontier becomes unattainable. An important property of second-best policies in dynamic economies is that they are generally not time-consistent or sub-game perfect. Adding the requirement of sub-game perfection along with incentive compatibility restricts the economy to a "third-best" efficiency frontier, which is *inside* the second-best one, wherever self-selection constraints bind. Not surprisingly, policies that might not have been sensible in a second-best world now become justifiable in a third-best one. These policies include many of the things that we observe in the real world such as in-kind transfers, quotas, minimum wages, rules for taxation-induced forced saving for retirement, various sorts of investment subsidies, and redistributional measures, such as subsidized drinking water, and the provision of free quasi-public goods such education and free medical care. In fact in the case of drinking water, an essential good for consumption and health, a negative consumption tax (i.e. a subsidy) is an obvious necessity. If any subsidy is at all justified, it is likely to be for drinking water. However, in the present low-tax climate it is unlikely to happen, although in some local jurisdictions in the US and also in Australia, equity considerations do seem to affect water-pricing decisions, as we shall see below in Sects. 5.7.1 and 5.7.3.

Nevertheless, the dominant trend in water pricing is that the water utility achieves at least complete cost recovery, including the recovery of capital costs, as the capital will have to be replaced when the capital equipment is worn out. Hence Ramsey pricing becomes of practical relevance. Below we continue the exposition of Ramsey pricing.

5.5.2 Ramsey Pricing Expressed as Covering Capital Costs

Ramsey pricing can be expressed in a variety of formulae all of which are equivalent. For the purpose of this chapter, we can outline Ramsey pricing by considering only two goods supplied by a public utility, that might receive a capital subsidy S, faces a capital cost K, and price P_i, marginal cost MC_i, and marginal revenue MR_i, for $i = 1, 2$.

Using this notation, we can summarize Ramsey as follows. Maximizing the sum of consumer and producer surplus, it can be shown that, at the constrained optimum:

$$\frac{P_1 - \mathrm{MC}_1}{P_2 - \mathrm{MC}_2} = \frac{\mathrm{MR}_1 - \mathrm{MC}_1}{\mathrm{MR}_2 - \mathrm{MC}_2} \tag{5.22}$$

subject to

$$\sum_{i=1}^{2} P_i Q_i = c(Q_1, Q_2) + K - S \tag{5.23}$$

where $c(Q_1, Q_2)$ is the total cost of production, so that the constraint represents total revenue equal to total cost plus capital costs (K), minus any capital subsidy (S), if positive. An equivalent statement of the optimum in Eq. (5.22) is:

$$\frac{P_1 - \mathrm{MC}_1/P_1}{P_2 - \mathrm{MC}_2/P_2} = \frac{E_2}{E_1} \tag{5.24}$$

subject to

$$\sum_{i=1}^{2} P_i Q_i = c(Q_1, Q_2) + K - S \tag{5.25}$$

where E_i is the price elasticity of user i.

In either formulation, P_i are the *Ramsey prices*. Formulation (5.24) is also known as the inverse elasticity rule. Suppose index $i = 1$ stands for the industrial users of water and $i = 2$ represents the residential users. If the elasticity of industrial users equals σ, it follows that

$$\sigma(P_1 - MC)/P_1 = (P_2 - MC)/P_2 \tag{5.26}$$

$(P_1 - MC)/P$ is a markup over MC required to cover average costs as well as the capital cost, net of any capital subsidy. Suppose $\sigma = 2$, i.e. the elasticity of industrial users is twice that of residential users, then the markup on industrial users will be half of that of residential users.

The difference between P_i and MC can also be viewed as the constrained optimal Ramsey commodity tax. If the subsidy S is zero, then the tax must fully cover all costs.

In order to apply Ramsey prices, certain assumptions must hold. These are:

(a) The public utility is using the least-cost state-of-the-art technology in drinking water production,
(b) Such technology is purchased in competitive markets,
(c) The capital-labor ratio in the public utility is optimal, i.e. the one that would hold in competitive labor markets, and
(d) All inputs, including labor inputs, are purchased in competitive markets.

If conditions (a)–(d) hold, then the public utility is said to be using the *best-practice* techniques. If any of the four conditions does not hold, then appropriate shadow prices must be computed to find the adjusted Ramsey prices. The adjusted Ramsey prices may be called the shadow Ramsey prices (*SRP*). The *SRP* will then be the benchmark for judging whether the prices that are being charged by a private sector partner are a *social improvement* with respect to public ownership or not.

In the empirical literature, it is often claimed that in Canada, the price of water is well below marginal cost. Renzetti estimates that "prices charged to residential and commercial customers are … only one-third and one-sixth of the estimated marginal cost for water supply and sewage treatment, respectively." (Renzetti 1999, p. 688). From this, Renzetti concludes that there is a significant welfare loss, associated with overuse of water. His findings can be illustrated in Fig. 5.1a. The purpose of the present chapter is not to determine if there is a welfare loss or not, but to concentrate on the best-practice technique for water production and estimate the Ramsey price, assuming one user. For the purposes of this paper, the benchmark is not the actual marginal cost, but the *shadow* Ramsey price.

Most water utilities in Canada and elsewhere have been relatively isolated from global water technology and for most water utilities the conditions (a)–(d) would be violated. This is because they have faced no pressure to cut costs or to adopt new technology (Brubaker 2011). Indeed many did not or could not (for a variety of reasons) make provisions for capital stock renewal. Most faced unionized labor, with the result that many facilities were grossly overstaffed. There are other monopolistic distortions, for which there is ample empirical evidence. The upshot is that actual market prices are much higher than *marginal social opportunity costs*. Shadow prices seek to "correct" for these distortions and estimate marginal social opportunity costs. Hence the actual price, though below actual marginal cost, could in fact be at or near the shadow Ramsey Price (SRP). This is illustrated in Fig. 5.1, parts (a) and (b). Section 5.6 below is an econometric estimation of Shadow Ramsey Prices. When taking Shadow Ramsey Prices into account, the statement that the water selling price is "too low" (Renzetti 1999) can be misleading; such statements do not take into account the theoretical results of the new public economics.

In Fig. 5.1a, the area EFG is Renzetti's welfare loss, due to the fact that actual price P^a is below P^b where P would equal current and actual MC, i.e. MC^a. In Renzetti's model, the breakeven Ramsey price (unadjusted) would be P^c, and optimal quantity supplied would be Q_6.

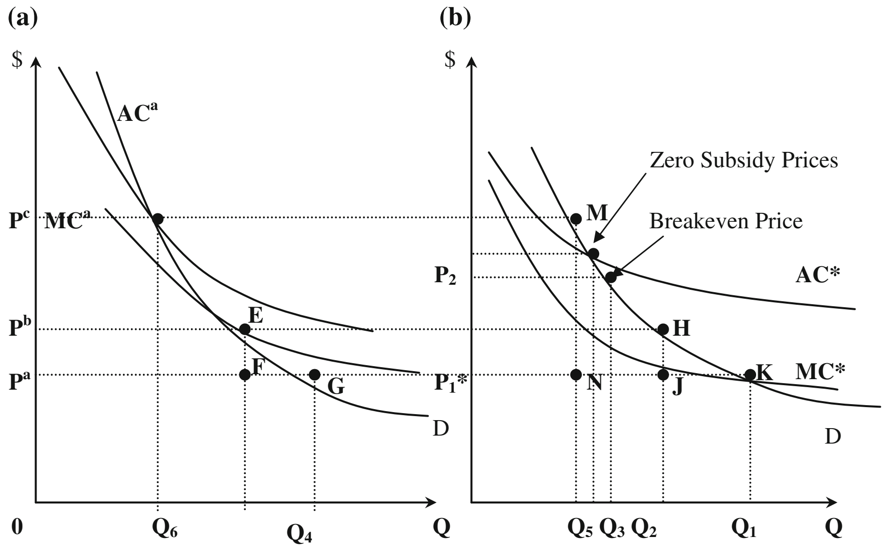

Fig. 5.1 The econometric estimation of Shadow Ramsey Prices

However if AC and MC are evaluated at the required shadow prices and therefore take into account market distortions due to monopolistic markets for capital goods, unionized labor, etc., then P^a may be close or even equal to P_1*, the Shadow Ramsey Price SRP^*, shown in Fig. 5.1b. In this case, imposing the price P^b, where price is equal to actual marginal cost MC^a would involve a welfare loss in the opposite direction. This loss is the area HJK in Fig. 5.1b. Moreover, imposing a Breakeven Price P^c in Fig. 5.1a would result in a very large welfare loss equal to the area MNK.

The observations above yield the following conclusions:

1. Just because actual (current) price is below actual MC does not necessarily mean that there is a welfare loss.
2. The policy conclusion from Fig. 5.1a, that price should be raised at least to P^b is also not valid.

A more *general* conclusion is that the priority when dealing with public utilities should be cost reduction through better technology, better management, and a reduction in labor input, rather than the price increase. This general conclusion is in line with the objectives of the legislation of the Ontario government that requires water utilities to become independent financial entities and raise revenues to cover all of their costs, including capital costs. The Ontario Clean Water Agency (OCWA) was created on November 15, 1993 under the Capital Investment Plan Act with a mandate to provide reliable water and wastewater services to Ontario municipalities on a cost-recovery basis. OCWA operates 450 facilities in the province, making it one of the largest operators of water and wastewater facilities in Canada (OCWA 2011).

5.5.3 Ramsey Pricing and Equity Issues

The theoretical discussion has been hitherto confined to achieving productive efficiency. The theoretical foundation of the above discussion is based on Ramsey pricing. Yet, the Ramsey model can incorporate not only efficiency considerations but also equity issues. In fact, Frank Ramsey presented his result as a theorem on taxation rather than pricing. The Ramsey optimum tax rule, that is, the percentage reduction in quantity demanded of each commodity be the same, was interpreted by Kahn (1970) as the inverse elasticity rule. The inverse elasticity rule states that the optimum tax rates and price elasticities of demand should be inversely related.

The Ramsey differential between price and marginal cost can be interpreted as an Optimal Commodity Tax. While for efficiency the optimal tax is determined by the inverse elasticity rule, there is no *a priori* reason why that tax should be the same for all income classes. Indeed, distributional considerations can be introduced depending on the "inequality averseness" of a particular country. The optimal tax literature shows that differentiated commodity taxes can be used to supplement income taxes as a redistributive tool (Atkinson and Stiglitz 1976). Wiegard (1980) shows that for an economy, uniform commodity taxes are optimal only if efficiency is the sole concern. On the other hand, once equity is introduced, the optimal taxes change drastically. These economists also demonstrate that taxes on basic necessities (such as food, electricity and housing) decrease and may even become negative. Of course water is a necessity and many nations have also accepted access to water as a human right.[1] The optimal commodity tax would depend on inequality averseness and so should not fall on necessities. In fact taxes on necessities could even be negative, whereas taxes on luxury goods such as restaurant meals, gasoline, communication goods, and personal equipment could rise.

Thus distributional considerations do not rule out a flat rate charge for water up to some threshold followed by a steeply rising price schedule. When taking distributional concerns into consideration, this could in fact override the inverse elasticity rule, under which industrial users pay a lower Ramsey price for water than residential users. But the actual post distributional taxes could introduce a much differentiated price structure for water. For example, water prices for low-cost housing units could be set below MC up to some threshold determined by health considerations.

[1] On 28 July 2010, through Resolution 64/292, the United Nations General Assembly explicitly recognized the human right to water and sanitation and acknowledged that clean drinking water and sanitation are essential to the realization of all human rights.

5.6 Econometric Estimation of Shadow Ramsey Prices

Before we proceed with the estimations, the following should be noted:

- We must distinguish between Short Run Marginal Costs (SRMC) and Long Run Marginal Costs (LRMC). In the short run the plant size is fixed, whereas the LRMC is dTC/dQ, in which plant size is variable. This is the Ramsey Price; when it is adjusted for market distortions, it becomes the "Shadow Ramsey Price."
- The SRMC is the cost of treating 1 extra cubic meter of water. This is equal to the energy cost of Reverse Osmosis.
- We have Fixed Costs (FC) + Variable Costs (VC) in the long run. The plants' borrowing rate for amortization is used to find a price at which TC = TR. This is the Breakeven Price.

To repeat, we can think of Ramsey prices as *long run marginal cost prices*. But integrating Ramsey prices with the new public economics would require *Shadow Ramsey Prices*. That is, for the public sector, it is appropriate to take into account the price distortions that we find in the real economy, distortions due to monopolistic economic structures. Hence we need to correct for these distortions. As an illustration, all actual real cost data are reduced by 2 percent to reflect the estimated shadow price (for the theory of shadow pricing, see Dreze and Stern (1990) and Little and Mirlees (1974). For those who wish to see "unadjusted prices," please multiply the estimated Shadow Ramsey prices by a factor of 1.02 . Details of shadow pricing techniques are outside the scope of this work. We show the econometric estimation of Shadow Ramsey Prices for 36 Reverse Osmosis technology water treatment plants.

We use a log-linear model, adjusted to remove heteroscedasticity, estimated by Weighted Least Squares (WLS), reported in Eq. 5.27a, b; and in Table 5.1 below. We distinguish between desalination of seawater and brackish water using the dummy variable "SEA." For brackish water, SEA = 0. In addition, the interaction term combining the effects of Plant Size "Q" and the dummy variable "SEA," was estimated and named "QSEA." On the basis of the economic theory, the double-log regression model is determined as the best. In Eq. 5.27a, b, TC is total cost, and Q is plant size.

$$\mathrm{In}\,\mathrm{TC}_i = \alpha + \beta \cdot \mathrm{In}\,Q_i + \gamma \cdot \mathrm{SEA}_i + \delta \cdot \mathrm{In}\,\mathrm{QSEA}_i + \mu_i \tag{5.27a}$$

$$\begin{aligned}\mathrm{In}\,\mathrm{TC}_i = {} & 13.885652232 + 0.740621837 \cdot \mathrm{In}\,Q_i + 0.928025419 \cdot \mathrm{SEA}_\mathrm{i} \\ & + 0.143438118 \cdot \mathrm{In}\,\mathrm{QSEA}_i\end{aligned} \tag{5.27b}$$

$$\text{Standard errors}: (0.291654726) \quad (0.18230551) \quad (0.352579975) \quad (0.188792389)$$
$$\mathrm{T-statistics}: (47.60990) \quad (4.06153) \quad (2.63210) \quad (0.75977)$$

Table 5.1 Adjusted double-log regression model (including dummy and interaction terms)

n	d.f.	R^2	$R(bar)^2$	SEE/SER	Durbin-Watson statistic
36	32	0.998883	0.998779	0.782302197	2.053142

n is the number of observations
d.f. are the degrees of freedom
SEE is the Standard Error of the Estimate

The model contains no heteroscedastic errors or heteroscedasticity and provides an exceptionally good fit to our dataset. We can now be confident to use this model for the estimation of Marginal Costs and Shadow Ramsey Prices.

5.6.1 *Derivation of MC for Two Types of Desalination*

This section uses the adjusted double-log model presented in Eq. 5.27a, b above and estimates the MC functions for both brackish water and seawater desalination plants. The coefficient estimators used are summarized in Table 5.2.

First, the total cost (TC) function obtained is simplified as follows:

$$e^{\text{In } T\hat{C}y} = e^{\hat{\alpha}} \cdot e^{\hat{\beta}\cdot \text{In } Q} \cdot e^{\hat{\gamma}\cdot \text{SEA}} \cdot e^{\hat{\delta}(\text{In QSEA})}$$

$$T\hat{C} = e^{\hat{\alpha}} \cdot Q^{\hat{\beta}} \cdot e^{\hat{\gamma}\cdot \text{SEA}} \cdot e^{\hat{\delta}(\text{In QSEA})}$$

$$T\hat{C} = e^{\hat{\alpha}} \cdot Q^{\hat{\beta}} \cdot e^{\hat{\gamma}\cdot \text{SEA}} \cdot \left[e^{\text{In } Q}\right]^{\hat{\delta}\cdot \text{SEA}}$$

This yields the following Total Cost function:

$$T\hat{C} = e^{\hat{\alpha}} \cdot Q^{(\hat{\beta}+\hat{\delta}\cdot \text{SEA})} \cdot e^{\hat{\gamma}\cdot \text{SEA}}$$

On a priori grounds, it makes good economic sense to separate seawater desalination from brackish water desalination. Accordingly, two Marginal Cost equations are derived from the TC function above.

For a *Brackish* water desalinating plant:

$$\{\text{SEA} = 0\}$$

$$\therefore T\hat{C} = e^{\hat{\alpha}} \cdot Q^{\hat{\beta}}$$

$$\therefore MC = e^{\hat{\alpha}} \cdot \hat{\beta} \cdot Q^{(\hat{\beta}-1)}$$

Using the coefficient estimates obtained in the adjusted log-linear regression model, we get:

Table 5.2 Estimated coefficients of the adjusted log-linear model

α	13.885652232
β	0.740621837
γ	0.928025419
δ	0.143438118

$$\mathrm{MC} = e^{13.885652232} \cdot 0.740621837 \cdot Q^{(0.740621837-1)}$$
$$\mathrm{MC} = e^{13.885652232} \cdot 0.740621837 \cdot Q^{(-0.259378163)}$$
$$\underline{\underline{\mathrm{MC}}} = \underline{\underline{1072660.1386.Q^{(-0.259378163)}}}$$

For a *Seawater* desalinating plant the equation is different:

$$\{\mathrm{SEA} = 1\}$$
$$\therefore T\hat{C} = e^{\hat{\alpha}} \cdot Q^{(\hat{\beta}+\hat{\delta})} \cdot e^{\hat{\gamma}}$$
$$TC = e^{\hat{\alpha}+\hat{\gamma}} \cdot Q^{\hat{\beta}+\hat{\delta}}$$
$$\therefore \mathrm{MC} = e^{\hat{\alpha}+\hat{\gamma}} \cdot \left(\hat{\beta} + \hat{\delta}\right) \cdot Q^{(\hat{\beta}+\hat{\delta})-1}$$

Using the coefficient estimates obtained:

$$\mathrm{MC} = e^{13.885652232+0.928025419} \cdot (0.740621837 + 0.143438118) \cdot Q^{(0.740621837+0.143438118-1)}$$
$$\mathrm{MC} = e^{14.813677651} \cdot (0.884059955) \cdot Q^{(-0.115940045)}$$
$$\mathrm{MC} = 2713304.0341 \cdot (0.884059955).Q^{(-0.115940045)}$$
$$\underline{\underline{\mathrm{MC}}} = \underline{\underline{2398723.4423 \cdot Q^{-0.115940045}}}$$

Finally, we can compute the Marginal Cost values for Seawater and Brackish water desalination. The 36 fitted values are computed below, using a standard econometrics program such as WinRats. Results are organized to increase with plant size and interpreted in Table 5.3. Furthermore, Fig. 5.2 illustrates Marginal Costs in brackish water and seawater desalination plants, respectively.

5.6.2 Derivation of Shadow Ramsey Prices and Breakeven Prices

In this section, the units of the optimal log-linear regression model are converted from US gallons to cubic meters. On the basis of the model expressed in cubic meters, Shadow Ramsey Prices and Breakeven Prices are computed.

In order to express MC, Shadow Ramsey Prices and Breakeven Prices in cubic meters, the constant coefficient α needs to be recalculated. Other coefficients in the model do not change. New coefficients are reported in Table 5.4.

The Shadow Ramsey prices (SRP) *follow* directly from the estimated Marginal Costs. The Shadow Ramsey Price for each capacity is the price where P = LRMC. Therefore, a *brackish* water desalinating plant faces the following Ramsey Price curve:

Table 5.3 Summary of computed MC for various sizes of RO desalination plants

	Dataset column 1	Dataset column 2	Dataset column 3	Dataset column 4
Variable name	TCy	Q	MC seawater	*MC brackish*
Units	[TC in US$/year]	[MGD or millions of gallons per day]	$/MG	$/MG
1	48983	0.01	4091290.05	2623100.81
2	95630	0.02	3775364.07	2191464.17
3	86793.35	0.04	3483833.63	1830854.22
4	360620	0.1	3132710.22	1443566.58
5	289080	0.13	3038852.54	1348597.66
6	221628	0.26	2804194.9	1126683.13
7	1045360	0.4	2667579.44	1007570.87
8	2628000	0.45	2631399.19	977254.67
9	6022500	1	2398723.44	794435.52
10	5767000	1	2398723.44	794435.52
11	2701000	1	2398723.44	794435.52
12	609550	1	2398723.44	794435.52
13	3321500	1	2398723.44	794435.52
14	578160	1.06	2382573	782518.95
15	3066000	1.2	2348550.5	757740.97
16	2581280	2.6	2147178.5	620045.6
17	2903940	2.6	2147178.5	620045.6
18	30700000	3	2111848.31	597453.1
19	7270800	3	2111848.31	597453.1
20	1543950	3	2111848.31	597453.1
21	11607000	5	1990405.51	523312.49
22	46532500	5	1990405.51	523312.49
23	13997750	5	1990405.51	523312.49
24	2427250	5	1990405.51	523312.49
25	11785000	5	1990405.51	523312.49
26	56200000	6	1948773.16	499140.96
27	85400000	7	1914253.61	479577.37
28	22009500	10	1836708.07	437200.34
29	4489500	10	1836708.07	437200.34
30	12148660	10.6	1824341.64	430642.32
31	190700000	14	1766436.55	400661.92
32	22356250	25	1651592.69	344717.67
33	54385000	25	1651592.69	344717.67
34	11041250	25	1651592.69	344717.67
35	81221625	64.5	1479720.68	269586.65
36	102200000	70	1465748.4	263924.99

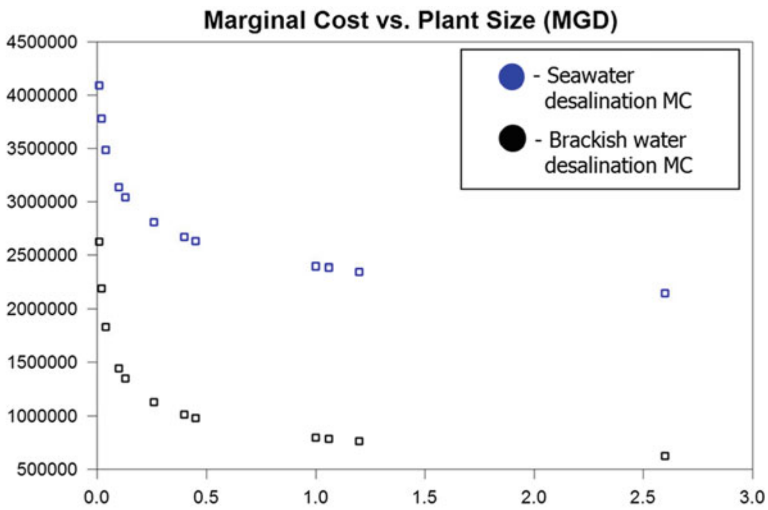

Fig. 5.2 Marginal cost versus plant size (MGD)

Table 5.4 Estimated MC, Shadow Ramsey Prices and Breakeven Prices in cubic meters

α	7.985754878
β	0.740621837
γ	0.928025419
δ	0.143438118

Note only the constant changes

$$\text{LRMC} = \text{Shadow Ramsey Price} = \hat{\beta} \cdot e^{\hat{\alpha}} \cdot Q^{(\hat{\beta}-1)}$$

Taking in the estimated coefficients,

$$\underline{\underline{\text{Shadow Ramsey Price} = 0.740621837 \cdot e^{7.985754878} \cdot Q^{(-0.259378163)}}}$$

On the other hand, a *seawater*-desalinating plant faces the following Shadow Ramsey Price equation:

$$\text{LRMC} = \text{Shadow Ramsey Price} = e^{\hat{\alpha}+\hat{\gamma}} \cdot (\hat{\beta} + \hat{\delta}) \cdot Q^{(\hat{\beta}+\hat{\delta}-1)}$$

When we include estimated coefficients, we get:

$$\underline{\underline{\text{Shadow Ramsey Price} = e^{8.913780297} \cdot 0.884059955 \cdot Q^{-0.115940045}}}$$

The computed Shadow Ramsey Prices are displayed in Table 5.5.

Table 5.5 Summary of computed MC for various sizes of RO desalination plants

	Q	Ramsey price brackish	Ramsey price seawater	Breakeven price brackish	Breakeven price seawater
Variable name	Plant size	Shadow Ramsey price	Shadow Ramsey price	Breakeven price	Breakeven price
Units	MGD [millions of US gallons per day]; (m^3/day)	US$/ brackish m^3	US$/ seawater m^3	US$/ brackish m^3	US$/ seawater m^3
1	0.01 (38)	1.8985	2.9612	2.5635	3.3495
2	0.02 (76)	1.5861	2.7325	2.1416	3.0909
3	0.04 (151)	1.3251	2.5215	1.7892	2.8522
4	0.1 (379)	1.0448	2.2674	1.4107	2.5648
5	0.13 (492)	0.9761	2.1995	1.3179	2.4879
6	0.26 (984)	0.8155	2.0296	1.1011	2.2958
7	0.4 (1514)	0.7293	1.9307	0.9847	2.1839
8	0.45 (1703)	0.7073	1.9046	0.955	2.1543
9	1 (3785)	0.575	1.7361	0.7764	1.9638
10	1 (3785)	0.575	1.7361	0.7764	1.9638
11	1 (3785)	0.575	1.7361	0.7764	1.9638
12	1 (3785)	0.575	1.7361	0.7764	1.9638
13	1 (3785)	0.575	1.7361	0.7764	1.9638
14	1.06 (4012)	0.5664	1.7245	0.7647	1.9506
15	1.2 (4542)	0.5484	1.6998	0.7405	1.9228
16	2.6 (9842)	0.4488	1.5541	0.6059	1.7579
17	2.6 (9842)	0.4488	1.5541	0.6059	1.7579
18	3 (11356)	0.4324	1.5285	0.5839	1.729
19	3 (11356)	0.4324	1.5285	0.5839	1.729
20	3 (11356)	0.4324	1.5285	0.5839	1.729
21	5 (18927)	0.3788	1.4406	0.5114	1.6295
22	5 (18927)	0.3788	1.4406	0.5114	1.6295
23	5 (18927)	0.3788	1.4406	0.5114	1.6295
24	5 (18927)	0.3788	1.4406	0.5114	1.6295
25	5 (18927)	0.3788	1.4406	0.5114	1.6295
26	6 (22712)	0.3613	1.4105	0.4878	1.5955
27	7 (26497)	0.3471	1.3855	0.4687	1.5672
28	10 (37853)	0.3164	1.3294	0.4273	1.5037
29	10 (37853)	0.3164	1.3294	0.4273	1.5037

(continued)

Table 5.5 (continued)

	Q	Ramsey price brackish	Ramsey price seawater	Breakeven price brackish	Breakeven price seawater
30	10.6 (40124)	0.3117	1.3204	0.4208	1.4936
31	14 (52994)	0.29	1.2785	0.3916	1.4462
32	25 (94633)	0.2495	1.1954	0.3369	1.3522
33	25 (94633)	0.2495	1.1954	0.3369	1.3522
34	25 (94633)	0.2495	1.1954	0.3369	1.3522
35	64.5 (244152)	0.1951	1.071	0.2635	1.2114
36	70 (264971)	0.191	1.0609	0.2579	1.2

The Breakeven Prices equal ATC. TC function was shown to be:

$$T\hat{C} = e^{\hat{\alpha}} \cdot Q^{(\hat{\beta}+\hat{\delta}\cdot \mathrm{SEA})} \cdot e^{\hat{\gamma}\cdot \mathrm{SEA}}$$

For a *brackish* water desalinating plant, Breakeven Prices are derived as follows:

$$\{\mathrm{SEA} = 0\}$$
$$\therefore T\hat{C} = e^{\hat{\alpha}} \cdot Q^{\hat{\beta}}$$
$$\therefore \mathrm{ATC} = e^{\hat{\alpha}} \cdot Q^{\hat{\beta}-1}$$

Using the coefficient estimates obtained in the adjusted log-linear regression model, we get:

$$\mathrm{ATC} = e^{7.985754878} \cdot Q^{(0.740621837-1)}$$
$$\underline{\underline{\mathrm{ATC}}} = \underline{\underline{e^{7.985754878} \cdot Q^{-0.259378163}}}$$

For a *seawater* desalinating plant, the Breakeven Price is higher:

$$\{\mathrm{SEA} = 1\}$$
$$\therefore T\hat{C} = e^{\hat{\alpha}} \cdot Q^{\left(\hat{\beta}+\hat{\delta}\right)} \cdot e^{\hat{\gamma}}$$
$$TC = e^{\hat{\alpha}+\hat{\gamma}} \cdot Q^{\hat{\beta}+\hat{\delta}}$$
$$\therefore \underline{\underline{\mathrm{ATC} = e^{\hat{\alpha}+\hat{\gamma}} \cdot Q^{\hat{\beta}+\hat{\delta}-1}}}$$

Using the coefficient estimates obtained:

$$\mathrm{ATC} = e^{(7.985754878+0.928025419} \cdot Q^{(0.740621837+0.143438118-1)}$$
$$\underline{\underline{\mathrm{ATC}}} = \underline{\underline{e^{8.913780297} \cdot Q^{-0.115940045}}}$$

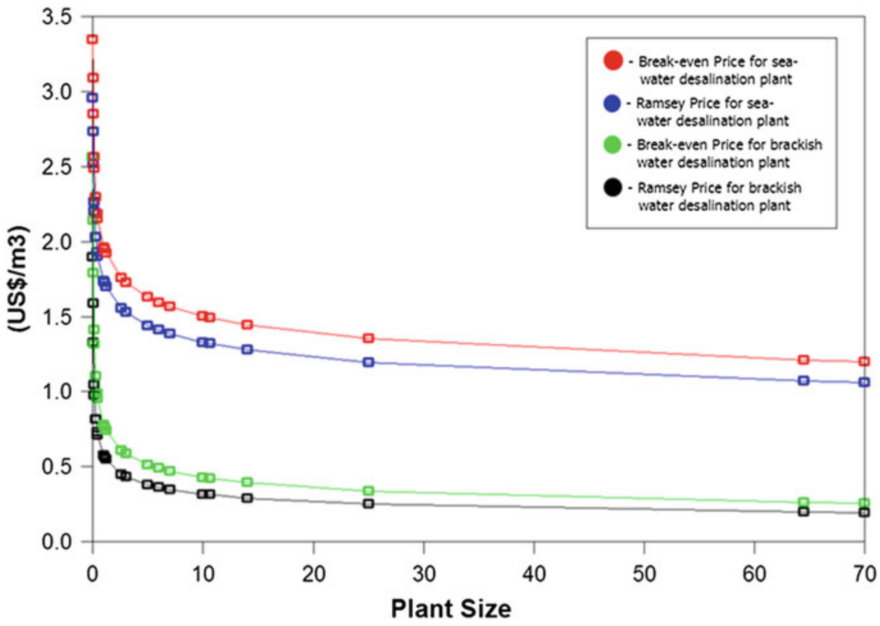

Fig. 5.3 Brackish and seawater desalination: Ramsey Price, Breakeven Price versus plant size

Table 5.5 summarizes the calculations of Shadow Ramsey Prices and Breakeven Prices for different plant capacities. Plant sizes are ordered to increase from the lowest plant capacity to the highest.

The objective of this subsection was to illustrate the computation of the Shadow Ramsey Prices per unit and also the Breakeven Price per unit, where there are economies of scale. We carried out the computations of the 36 reverse osmosis treatment plants shown in Chap. 4.

The data from Table 5.5 is finally plotted in the next three (Figs. 5.3, 5.4 and 5.5). Figures 5.4 and 5.5 simply break up the capacity ranges; Fig. 5.4 is for capacities between 0 and 3 m^3 and Fig. 5.5 is for capacities 3–70 m^3.

To summarize, there are several pricing concepts: (a) short run marginal cost is ∂TC/∂VC, (where VC is Variable Cost) holding plant size constant, which is essentially the energy cost of producing one more unit of water through reverse osmosis; long run marginal cost or Ramsey Price is dTC/dQ, where capacity Q is variable; Shadow Ramsey Price is α.dTC/dQ, where α is a multiplier correcting for market imperfections; and Breakeven cost is price where Total Revenue TR = TC, or just the average total cost ATC = TC/Q.

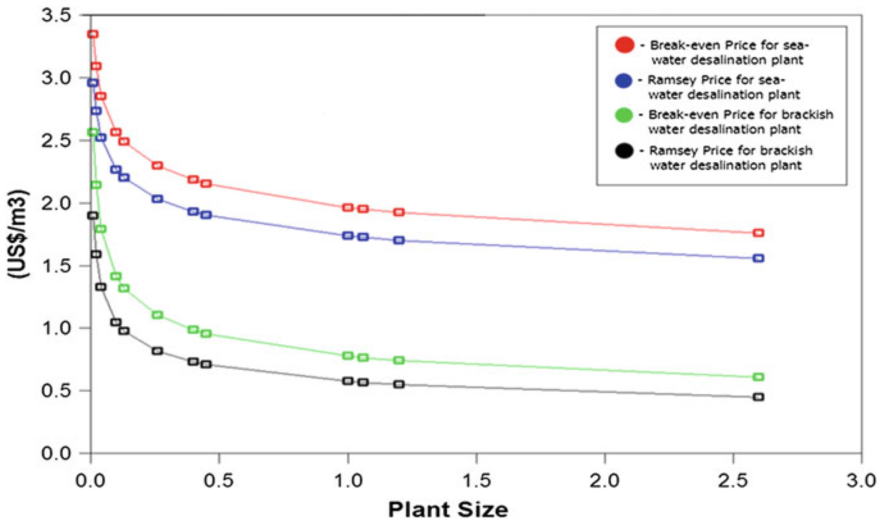

Fig. 5.4 Brackish and seawater desalination: Ramsey Price, Breakeven Price versus plant size (0 < Q < 3)

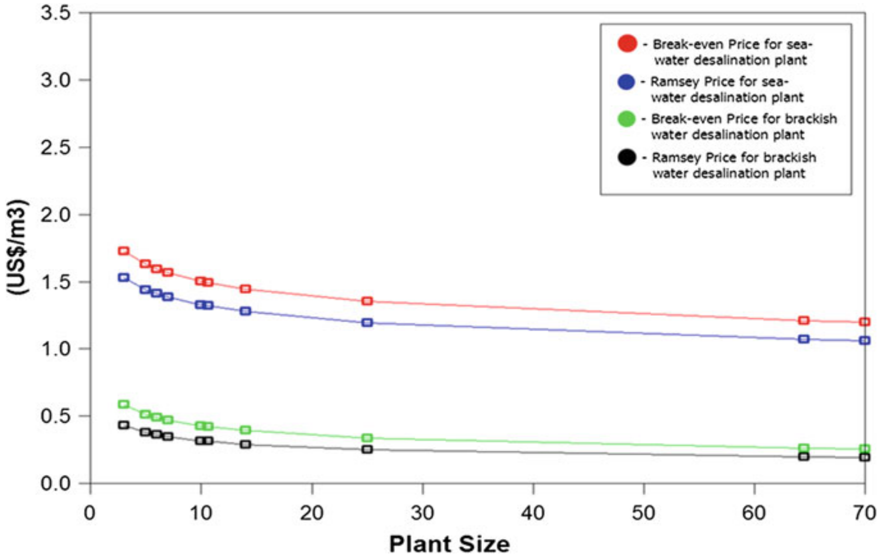

Fig. 5.5 Brackish and seawater desalination: Ramsey Price, Breakeven Price versus plant size (3 < Q < 70)

5.7 Water Pricing in Developed Countries

5.7.1 Water Pricing Practice in the US

Currently, there are roughly 50,000 community drinking water systems in the US and most of these systems are owned by municipalities (USEPA 2009). Privately owned systems merely account for 8.3 percent of total water supply, and a state public utility commission almost invariably regulates the privately owned systems. Only a few state commissions have no authority over the water sector (e.g. in Georgia, Michigan, Minnesota, North Dakota, South Dakota, and Washington) (Beecher 2011).

The US water systems have properly followed the guidelines in American Water Works Association) Manual M1, Principles of Water Rates, Fees, and Charges. As a manual of standard practice, the American Water Works Association advocates "the use of the generally accepted cost-based principles and methodologies for establishing rates, charges, and fees…" to "provide sufficient funding to allow communities to build, operate, maintain, and reinvest in their water system that provides the community with safe and reliable drinking water and fire protection" (Zieburtz and Giardina 2012). Thus, as an important component in a well-managed and operated water system in the US, the "cost-based rates, fees, and charges" principle is applied. Moreover, all the numerical examples provided in Manual M1 are illustrative and the pricing methods may vary depending on the specific local conditions.

Water charges for each customer class mainly consist of two rate components: a fixed charge that may differ among the meter sizes and a volumetric consumption charge (i.e. uniform rate, increasing-block rate or decreasing-block rate) depending on water consumption. The fixed costs associated with infrastructure, distribution network, service and fire protection are normally recovered through fixed charges, while the volumetric-related costs cover the rest of the water system's annual costs. That is, the additional costs are commonly recovered by a volumetric consumption charge (Zieburtz and Giardina 2012). Many municipal water utilities follow the practice of recovering the distribution-related costs by volumetric charge; but in many cases, this probably leads to revenues being inadequate to cover distribution costs alone. Overcast (2012), a director in the Ratemaking and Financial Planning Services Group at Black and Veatch, proposed that fixed distribution-related costs be recovered by a fixed service charge. Moreover, all the water systems seem to be facing declining per capita consumption as a result of a successful conservation-oriented policy. If the volumetric charges are raised, revenue tends to drop in the future and this exacerbates the problem of recovering fixed costs since high-volume water users usually have an elastic demand (Zieburtz and Giardina 2012). Furthermore, the pressure on water costs is also due to the substantial fixed costs associated with replacing the critical infrastructure.

A recent survey conducted by Beecher and Kalmbach (2013) focused on the water pricing practices in eight states located in the Great Lakes Water Basin:

Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania, and Wisconsin. In particular, 10 top water systems (based on the service population) in each state are included in the survey. The major findings are summarized below (Beecher and Kalmbach 2013):

(a) Water charges are considerably lower for municipal water systems and higher for private water systems (Table 5.6). Comparatively higher charges for privately owned systems are associated in part with income taxes and the cost of equity, but possibly with higher territorial service costs as well as various costing and ratemaking practices. In addition, other revenue sources of municipal water systems such as low-cost debt financing or government grants and transfers could be another explanation. Compared to municipal water systems, the private systems always impose fixed charges to cover the cost of fire protection and other service costs. Furthermore, the scale economies are not apparent, especially for private water systems; in fact diseconomies associated with distribution-related costs and other costs have been a concern for both municipal and private water systems.

(b) Some 76 percent of the water systems primarily rely on surface water, which imposes higher water charges compared to the systems relying on groundwater, mainly because higher costs are associated with meeting surface water treatment requirements set by the USEPA.

(c) The decreasing-block rates are more common for nonresidential consumers, while increasing-block rates are more often used for residential consumers (see Table 5.7).

(d) In order to improve water efficiency, most water systems provide conservation information and tips to the customers they serve. In addition, some water systems take fairness and ability to pay into account through discounts for low-income households and seniors (Table 5.8).

(e) The fixed charge in water prices varies by the quantity of water supplied and in particular the lower fixed charge is always for higher-volume water usage (see Fig. 5.6).

Table 5.6 Average charges for monthly water usage (Beecher and Kalmbach 2013)

Water systems	Residential			Commercial		Industrial	
Meter size	5/8″	5/8″	5/8″	5/8″	2″	4″	8”
Water consumption	0 cf	500 cf	100 cf	3000 cf	50,000 cf	1,000,000 cf	2,000,000 cf
Municipal (56)	$6.29	$14.58	$24.96	$65.40	$966.00	$17,087.00	$33,773.00
Private (5.13)	$14.66	$34.71	$55.13	$128.93	$1,613.00	$25,171.00	$47,849.00
Surface water (61)	$8.00	$19.33	$32.50	$84.04	$1,192.00	$20,342.00	$39,784.00
Groundwater (5.19)	$7.71	$16.21	$26.86	$63.73	$860.00	$14,626.00	$28,842.00

Note cf stands for cubic feet

Table 5.7 Water pricing structure for residential and nonresidential consumers (Beecher and Kalmbach 2013)

Basic rate structures	Residential	percent	Nonresidential	percent
Decreasing-block	35	44	48	60
Uniform	30	38	25	31
Increasing-block	14	18	5	6
Combined-block	1	1	2	3
Total	80	100	80	100

Table 5.8 Conservation and assistance policies (Beecher and Kalmbach 2013)

	Number of systems	percent
Conservation information/tips	57	71
Payment assistance	21	26
Low-income discount	9	11
Senior discount	8	10

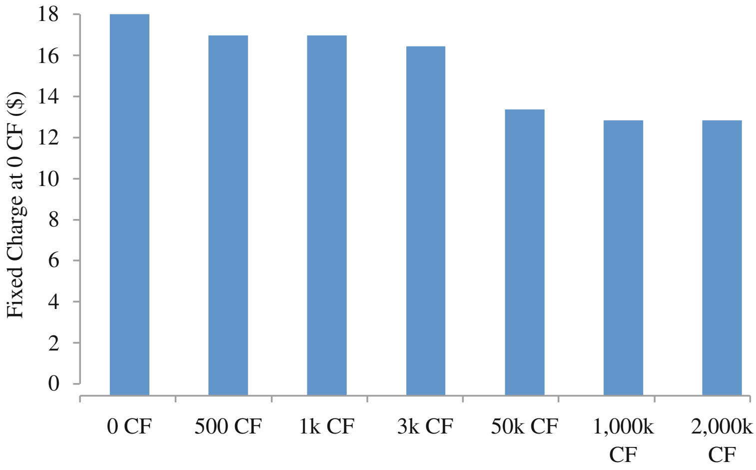

Fig. 5.6 Fixed charges vary by the quantity of water usage in cubic feet (CF) (Reproduced from Beecher and Kalmbach 2013)

(f) Finally, Beecher and Kalmbach pointed out "pressure on water prices is due to the movement toward cost-based and more efficient prices, exacerbated by historical underpricing by some nonprivate systems, loss of subsidies and transfers, and flat or declining demand."

The overall picture shows that for the US as a whole, water is typically publicly provided by a local municipality. All water systems have prices that cover full capital and operating costs. About 20 percent of the utilities in the Great Lakes Basin sample also take into account ability to pay and offer discounts to seniors and those with low incomes. This is consistent with the theory of the new public economics.

5.7.2 Water Pricing Practice in the European Union

In 2000, the "Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000, establishing a framework for Community action in the field of water policy" (EC Water Framework Directive 2000/60/EC) was released and came into force in the same year. The purpose of this Directive was to "coordinate Member States' efforts to improve the protection of Community waters in terms of quantity and quality, to promote sustainable water use, to contribute to the control of transboundary water problems, to protect aquatic ecosystems, and terrestrial ecosystems and wetlands directly depending on them, and to safeguard and develop the potential uses of Community waters" (EC Water Framework Directive 2000/60/EC). It is important to note that the Directive 2000/60/EC recommends all EU countries achieve full cost recovery in pricing water. According to Article 9 of Directive 2000/60/EC, (a) member states shall follow the principle of "recovery of the costs of water services"; (b) the water-pricing policies need to provide adequate incentives for users to use water resources efficiently by 2010; (c) member states shall implement policies that are in accordance with the principle of "polluter pays," (d) environmental-related costs (i.e. damage to ecosystems being caused by pollution) and resource-related costs (i.e. over-abstraction of water sources in rivers, lakes, wetlands and aquifers) should be included in the total costs; and (d) the effects of economic, geographic and climatic factors on the costs shall be taken into account. Moreover, the EU Commission recommends a three-part tariff that includes (a) a fixed component to cover the fixed financial costs of supply, (b) a charge per unit of water used, and (c) a charge per unit of pollution produced. Furthermore, in order to assure drinking water safety, drinking water treatments are required to meet all the microbiological, chemical and organoleptic parametric standards under the EU Drinking Water Directive 98/83/EC. Each member state shall publish a report every three years on the quality of drinking water.

However, there are wide variations between EU member states in price levels, which could be caused by variations in costs, returns on capital, investment needs and sources, structure of prices, willingness to pay higher water prices, even who sets the prices, and so on (Hrovatin and Bailey 2001). Cost differentials are primarily due to the availability and proximity of water, environmental protection, and variations in the quality of drinking water. In general, the less urbanized member states in southern Europe face greater cost increases than more heavily urbanized northern European countries which have already paid for substantial capital costs to meeting the requirements of EU directives (Hrovatin and Bailey 2001). Moreover, some member states do not allow their utilities to make profits. For example, Denmark, Sweden, Belgium, and Ireland require utilities to break even, but make no profits. Furthermore, most EU member states (such as Denmark, Germany,

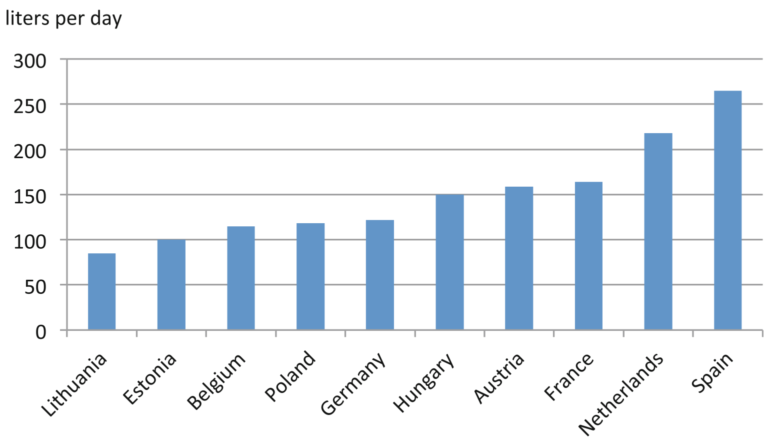

Fig. 5.7 Daily water use in selected EU countries in 2012 (liters per day) (Biswas and Kirchherr 2012)

Ireland, Hungary and Sweden) already use the two-part tariffs for the supply of water, while Greece and Turkey have applied the increasing-block tariffs. To show the variety of pricing practice, we present three case studies in EU member states: Ireland, France, and Spain.

In addition, water usages also widely differ among EU countries (see Fig. 5.7). In 2012, the average person in Spain used around 265 liters per capita per day, followed by the Netherlands with 218 per capita per day and France with 164 liters per capita per day. The more sustainable European water consumers are Lithuania, Estonia, and Belgium with 85, 100 and 115 liters per capita per day, respectively (Biswas and Kirchherr 2012). As noted by Biswas and Kirchherr, "these huge consumption differences are mainly due to different water pricing regimes across Europe that European policy makers have failed to harmonize despite the Directive. Pricing is the single most powerful policy tool to alter water consumption patterns and users' behavior.

5.7.2.1 Ireland

Ireland's water sector consists of local authorities and group water schemes. About 92 percent of Ireland's population is served by a total of 34 local authorities, while 8 percent of the population is served by group water schemes (Brady and Gray 2013). The pricing mechanisms for each local authority are different. In Ireland, residential charges have not been imposed since 1991 as full capital costs for the provision of water services to the residential sector are met by the Exchequer through indirect taxation, while the nonresidential sector is charged for water consumption and the charges are based on meter reading, including a fixed charge and a constant volumetric charge (Brady and Gray 2013). Ireland was ranked third cheapest in terms of the average cost of provision of water and wastewater services in the largest cities compared to other member states (Forfas 2008).

According to a report from the Local Government Efficiency Review Group in Ireland (Ireland Department of the Environment, Heritage and Local Government (DoEHLG) 2010) the costs of provision of water services have been increasing by roughly 5 percent per year since 2007, primarily due to (a) increases in energy costs, (b) regulatory compliance with both national and EU legislation, and (c) infrastructure investment costs for both water and wastewater treatment. Although the local authorities have been permitted to charge the full costs of provision for water and wastewater services to nonresidential users, full cost recovery has not been achieved (Forfas 2008). It should be noted that the water charges in Ireland do not include profit. Currently, the nonresidential sector is only charged the marginal cost of providing water services[2] (Commission on Taxation 2009). It is important to note that in addition to the cost associated with infrastructure renewal, operating costs are fully recovered in other EU member states (OECD 2010). Moreover, a lack of independent regulation of nonresidential charges leads to considerable pricing variations among the local authorities and the group water schemes (Brady and Gray 2013). A pricing assessment conducted in 2008 showed large pricing differences, with charges ranging from €0.99/m3 in Galway City to €2.71/m3 in Wexford. The average charge was €2.08/m3 (Forfas 2008). As a matter of fact, the volumetric charges imposed by group water schemes were some 35 percent lower than the prices charged by local authorities, resulting in inadequate cost recovery (Brady and Gray 2013).

5.7.2.2 France

Although some municipalities in France provide water and sewage services directly, most municipalities delegate the management of all or part of the public water supply utility to a private operator with contracts with a predetermined duration (Porcher 2014). This contract defines the payment to the operator, which will be included in the water price to be paid by the users. According to a 2006 report from *Dexia Crédit Local de France* (2006), 63 percent of French medium-sized cities contract out the services of potable water treatment and distribution, and 58 percent also contract out their sewage treatment. Moreover, 71 percent of the population in France is served by a private operator for water provision and 56 percent for water sewage (Cour des Comptes 2011). As a result, the unregulated private operators tend to maximize profit by pricing above marginal cost, resulting in a level of output below the socially optimal level (Porcher 2014). As shown above, a standard result is that social efficiency requires that marginal prices equal

[2] This marginal cost is the difference between the cost of providing water service infrastructure to residential users and the total cost of providing water to all users (Forfas 2008).

marginal costs. According to a source (Porcher 2014), residential customers in France face prices for water that average about 8 percent more than marginal costs.

Water pricing and household incomes in France vary from one municipality to another. The relatively high prices have been a concern for the municipalities with lower household incomes. During the period 1979–2005, the share of households' expenditures on water bills increased from 24 to 48 percent for the households in the lowest decile (i.e. the lowest 10 percent of the income scale). To deal with this issue, in 2000, the public and private operators developed a funding assistance program to subsidize those households who had financial difficulties in paying their water bills. As an alternative solution, Simon Porcher (2014) found that two-part tariffs[3] would be helpful to lower the fixed charges for poor households. That is, the fixed charges can vary depending on different classes of consumers.

5.7.2.3 Spain

The legal framework in Spain, Law 7/1985 on the Regulation of Local Government Terms and Conditions and Law 57/2003 on Local Government Modernization Measures, stipulates that local governments are responsible for urban water services, or they could choose how water services are managed within the legal framework. The legal regimes for the provision of municipal services are regulated under the Royal Decree 2/2000. As shown in Fig. 5.8, the local government can choose either in-house management[4] or outsourcing to an external company. In the latter case, full privatization or public–private partnership (PPP)[5] or partially privatized to a mixed company such as institutionalized PPP[6] can be considered. It is important to note that "Spanish legislation only contemplates privatizing the management of the service, as the infrastructure remains public property" (García-Valiñas et al. 2013)

The external involvement of public or private companies has been more widespread in Spain since the 1980s due to the implementation of more stringent legal requirements and the existence of a fragile financial situation in several municipalities (García-Valiñas et al. 2013). Moreover, privatization has been a source of

[3] A marginal cost pricing approach would use two-part tariffs with a price set to marginal cost and fixed charges equal to total fixed cost (Coase 1946). In the water industry, the two-part tariffs imply setting the fixed charges equal to each customer's share of the utilities' fixed costs and the volumetric charges equal to marginal costs (García-Valiñas et al. 2013).

[4] In-house management means that the local government provides the water service itself. The city council is responsible for decision making and management, uses its own employees and covers production costs with funds from the municipal budget (García-Valiñas et al. 2013).

[5] Contractual PPPs is a form of privatizing public services in Spain. That is a local government entrusts an individual or corporation to manage the urban water service but retains ownership (García-Valiñas et al. 2013).

[6] Institutionalized PPPs refers to the private sector participating in the management of the urban water service, while capital is shared between the private and public sector (García-Valiñas et al. 2013).

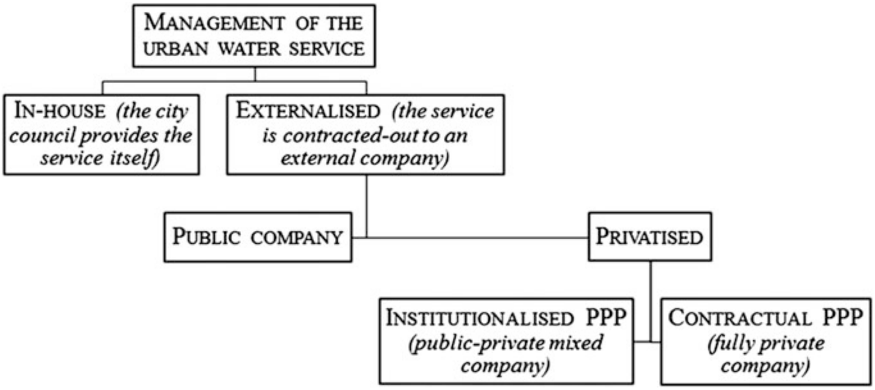

Fig. 5.8 Legal forms for the management of urban water services in Spain (García-Valiñas et al. 2013)

significant revenue for local governments (García-Valiñas et al. 2013). For example, Table 5.9 shows the relative share of different ownership regimes for the provision of urban water services in Andalusia (within Spain). Town councils provide water services to nearly 49 percent of Andalusian municipalities but only to 12 percent of the population, while public companies provide urban water services to 29 percent of the municipalities and to almost one half of the population in Andalusia. Contractual and institutionalized PPPs jointly provide urban water services to 23 percent of the municipalities and to almost 40 percent of the population of Andalusia. Accordingly, private utilities have established their business in medium and large-sized municipalities to pursue profits.

Even though a two-part tariff system is applied in most municipalities of Spain, water pricing varies greatly from one municipality to another due to lack of regulation. The findings from a recent case study in Andalusia demonstrated that the prices are lower when the urban water is directly provided by the public authority than when the service has been outsourced to the private sector (García-Valiñas et al. 2013). Once the service is outsourced, private companies set higher prices than municipalities due to larger fixed costs. Furthermore, institutionalized PPPs charge higher prices for water than contractual public–private partnerships.

It is unclear whether the Spanish water sector has reached compliance with the EU Water Directive; it is possible that Spain is in a transition phase.

5.7.3 Water Pricing Practice in Australia

The distribution of water and the provision of water services in Australia are managed by the public sector. In particular, local governments manage the provision of water services. Currently, most urban water utilities in Australia with successful water pricing schemes use some form of two-part tariff structure, which resulted from the 1994 Council of Australian Governments Water Reform Agreement (Australia National Water Commission 2011). As noted by Rogers et al. (2002),

> One of the main advantages of the two-part tariff system is the stabilized revenue base it affords the supplier. The fixed charge protects the supplier from demand fluctuations and reduces financial risks. The volumetric charge can vary according to the consumption level, [which] therefore encourages conservation.

In Australia, local governments and water businesses have set the volumetric tariff based on the long run marginal costs to provide signals for conservation and for efficient water use. Long run marginal costs can be defined as the "cost attributable to an extra permanent unit of consumption in bringing forward the future capital program" (Australia National Water Commission 2011).

Drought has always been a serious concern in Australia. According to the data from Australia National Water Commission (2011), the local governments and water businesses invested billions of dollars in supply augmentation such as desalination plants through "high cost, high reliability" water sources. For example, the total cost of desalination plants in Melbourne was $3.50 billion; in Sydney it was $1.83 billion; Perth, $1.34 billion; the Gold Coast, $1.20 billion; and Adelaide, $1.83 billion. The total investment of all these desalination plants amounts to $9.7 billion. Moreover, in addition to increasing volumetric charges to balance supply and demand in periods of drought, water authorities and governments in Australia have chosen to impose mandatory water restrictions to reduce consumption through an increasing-block tariff (Australia National Water Commission 2011). Thus if a household uses water above a given consumption threshold, it will pay a higher volumetric price for its water. It seems to encourage water conservation, but there is a disadvantage associated with an increasing-block tariff. For example, a poor household with a larger family may have higher water usage than a high-income household with a small family, and yet the large family, poor household would end up paying higher prices for water than the high-income household. Therefore, such pricing is inequitable.

As noted by the Australia National Water Commission in a recent review of water pricing reform (2011):

> Consumption-based or volumetric pricing led to demand reductions and more economically efficient water use. Through the recent drought, however, problems with the current approach to setting the volumetric tariff based on the long run marginal cost [LRMC] of augmenting supply were brought into sharp focus. LRMC prices ostensibly signalled the future costs of capacity augmentation to meet growth over the longer term but did not respond to increasing scarcity of water and did not reflect the high degree of variability in inflows. The problem with fixed pricing is it ignores the effect of weather on supplies. If there is a drought and reduced inflows into catchments and dams, then less water will be available. With fixed pricing, the amount charged to consumers is unchanged and the price is set too low to balance demand and supply in dry years.

The current urban water tariffs are therefore not responsive to changes in the value of water and thus lead to inefficient use (or nonuse) of water because the current approach to pricing does not reflect changes in short-term water availability (Australia National Water Commission 2011). Recently, the Australian Federal Government proposed a number of new pricing mechanism options (Australia National Water Commission 2008). For example, scarcity prices depend on the quantity of water in the dams. When the quantity of water available in the dams declines, the water prices are higher. Therefore, scarcity pricing significantly reduces the risk of revenue shortfalls for the water utilities when water in the dams is lower than expected. Moreover, as water prices rise, total sales of water tend to fall. In addition to improving efficiency and effectiveness in the provision of water services and water resource management, scarcity pricing reduces the need for additional bureaucracy that would be required to administer water restrictions and rationing (Australia National Water Commission 2008). However, Hunt et al. (2013) argued… "it is unclear how the scarcity pricing mechanism will enhance the transparency of information in accounting for the efficiency and effectiveness of sustainable water services…" This is because the performance of the pricing mechanism will depend on how the three levels of government (Federal, state and municipal) disseminate to the public the information about the pricing system. Hence inadequate communication with households may make scarcity pricing less effective.

5.8 Conclusions

In this chapter, we reviewed the classic theory of marginal cost pricing which comes to the conclusion that for social efficiency, public utilities should charge marginal cost for the public service or product such as water, and that the capital cost for water installations should be covered by general income and inheritance taxes, possibly by the central government, and that it is a fallacy to require that "every tub sit on its own bottom." Hotelling correctly predicted that the "rich" and the "land speculators" would object to marginal cost pricing. Frank Ramsey in 1927 showed that when capital costs must also be covered by the utility, then a price higher than marginal cost is second-best optimal, provided such prices are inversely proportional to the elasticity of demand, and that the lower the elasticity the higher the price. In fact Ramsey was showing what the optimal commodity tax should be and that it would follow the inverse elasticity rule. This second-best rule in general does not yield a "second best Pareto optimum," since one distortion cannot be completely offset by another. The new public economics has concluded that, in general, the second-best optimum is likely to be "inside" the first-best Pareto optimum frontier, but once dynamics are taken into account even the second-best optimum is unattainable, and that for public projects it would be necessary to evaluate the project at marginal social opportunity costs, which would require calculating "shadow prices," or market prices that are adjusted to arrive at marginal social opportunity costs. In this chapter, we called those prices the "Shadow Ramsey Prices." For the best water treatment technology, namely reverse osmosis,

Table 5.9 Share of different ownership regimes for the provision of urban water services in Andalusia (García-Valiñas et al. 2013)

	Municipalities (percent)	Population (percent)
In-house	48.80	11.60
Public company	27.80	49.60
Institutionalized PPP	10.40	15.10
Contractual PPP	13.00	23.70

we estimated Shadow Ramsey Prices to demonstrate that actual water pricing policy can indeed be integrated with the new public economics.

Equity and redistribution looms large in the new public economics and it is indeed not only feasible but possible to have a two-part water price such that for some minimum quantity of water required for health and survival, a zero or a lower price is justifiable, followed by a steeply rising price for outdoor water use. In a "third" best world, the water utility could do worse than utilize marginal cost pricing. In the water utility sector, this form of pricing is called "pricing based on cash needs." When we consider the details of "pricing based on cash needs," it is clear that it is precisely the same as marginal cost pricing. Of course accountants disapprove of pricing based on cash needs and political authorities tend to support the accounting point of view. For example, full cost accounting for water is the law in Ontario. In Alberta, marginal cost pricing survives in the form of "cash needs" in some small communities but this practice is due to be phased out as the government of Alberta also supports the accountants on how water should be priced. That is, water should be priced as in any other private business, based on full cost accounting.

Can water be under-priced? Yes it can, as when consumers pay a fixed annual charge irrespective of the amount of water consumed. In the UK, there were a number of areas without water meters and where water was distributed for a fixed charge. But that is now changing. By 2015, up to 92 percent of consumers in the south-east of England who get their water from "Southern Water" will be metered in line with the rest of Europe. From a public economics point of view, it can also be over-priced, when the price is way over marginal cost, but the higher price (in the public sector) may have a different social objective such as conservation. It is the need for conservation and reduction in waste that explains the high water prices in some northern European countries like Germany, Denmark, and the Netherlands. If the water utility is in the private sector, then pricing decisions will be governed by profit maximization, with no regard to social objectives.

From the point of view of the new public economics, equity and redistribution, as well as conservation can be social objectives driving pricing policy. We note that Australia uses Ramsey Pricing, as they rely on pricing water on the basis of long run marginal costs; in the US, there are jurisdictions that take equity issues into consideration; Germany and other northern European countries emphasize conservation and price water accordingly. Thus in the new public economics, social objectives are paramount; principles that are sacred to accountants are completely irrelevant from the point of view of economic theory.

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Part III
Incorporating Risk in Decision-Making

In Part III, we attempt to answer the following questions:

- In light of the recommendation of the World Health Organization (WHO), how can risks to health from drinking water be minimized?
- As risks to health can arise from water sources, treatment, and distribution, how can risk be characterized in source waters?
- What are the major threats to healthy and clean water?
- How can risks to health be minimized when the sources of contamination of source water are not known?
- What are the main dimensions of risk assessment in drinking water production?
- How can water utilities be organized so that risk can be minimized at the treatment stage?
- Are there hazards associated with the use of chlorine?
- What is the Hazard Assessment and Critical Control Points (HACCP) protocol and what is a "Water Safety Plan" and how can these be implemented?
- What is Quantitative Microbial Risk Assessment (QMRA), and how can it be used to enhance water safety?
- How can water utilities manage their large infrastructure assets so that health risks are minimized?
- What metals in pipes pose a major threat to long-term health?
- What can we learn about risk minimization from the WHO and from the protocols of the International Standards Organization (ISO)?

Chapter 6
Risk Assessment for Safe Drinking Water Supplies

6.1 Introduction

Potable water is considered as a public service, typically supplied by the local (municipal) public sector. The water supply authority is therefore a public "utility," such that it charges for water on the basis of being able to break even financially, whether or not it receives a partial subsidy from a higher jurisdiction. There is an economic justification for treating a water utility as a natural monopoly and hence as a "public utility" and not as a private business. There are strong economies of scale in water treatment and distribution. Consequently, if the water utility were in private hands, it would have to be regulated not just for health but also for the protection of consumers against monopolistic pricing. Water utilities in general have no profit motive and no incentive to improve water quality or to invest in more advanced water treatment technologies. In North America and also in Europe, there are pockets of privatized water companies, but there is no evidence of technological innovation in such companies and treated water quality is no better than that of the publicly owned utilities (Dore et al. 2004). In general, regulation and consumer awareness push water utilities to improve water quality and yet provide water at the lowest possible cost. In North America and many other countries around the world, the abundant availability of water from lakes and rivers makes surface water an easy option for water supplies, despite the fact that the quality of most surface water sources is degraded by point and nonpoint sources of pollution. The presence of pathogens and chemical contaminants in surface water is the major cause of many acute and chronic health diseases around the world. Source water quality degradation has led to a number of waterborne disease outbreaks in North America and elsewhere as documented in Chap. 2.

In the scientific literature, there is overwhelming evidence of health risk associated with untreated drinking water. But even in treated drinking water, there is a strong case for introducing some form of risk averseness and risk management. The objective of this chapter is to review and assess the major types of risk management

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methods for the three stages of potable water supplies; these are source water protection, water treatment, and pipeline distribution network. We also present a brief assessment of case studies where risk assessment methods have been applied.

This chapter is organized as follows. Section 6.2 covers principles of source water protection, for both point and nonpoint sources of pollution in greater detail than covered in Chap. 2. Section 6.3 covers the main approaches to risk assessment for potable water supplies. Section 6.4 provides an overview of some risk assessment case studies. Section 6.5 is a brief evaluation of the risk assessment process and includes suggestions about types of risk assessment that are likely to succeed.

6.2 Source Water Protection

Effective water source protection requires an understanding of the concept of watershed management and sources of water pollution (point and nonpoint) that contribute to degradation of source water within the watershed.

6.2.1 Principles of Watershed Management

Surface water quality is affected by surface runoff that receives a variety of pollutants. The application of watershed management is a useful tool for understanding and controlling pollution of surface waters. As stated in Chap. 2, a watershed is a land area, a bounded hydrological system, within which all flowing water bodies such as rivers and streams merge in one outlet. A large watershed can be formed from several smaller subwatersheds.

Developing a watershed management framework is necessary to minimize adverse environmental problems in advance. The framework describes the goals, outlines the protective actions, and focuses on a continued process for partners and stakeholders to work together in supporting the watershed management plans (USEPA n.d.). These partners and stakeholders make decisions on all aspects of the framework including (a) water quality standards, (b) watershed management approaches, and (c) individual management projects affecting localities and "areas of concern," where there is evidence of environmental stresses. Eventually, coordinated efforts in watershed management facilitate economic development and the implementation of environmental protection. Moreover, using sound science in watershed management helps to achieve sustainable goals, such as environmental cleanup or even relieving the pressure of water demand due to increasing urban populations. A watershed is affected by a variety of factors that include geology, hydrology, soils and vegetation, land use, and climate (USEPA n.d.). Furthermore, watershed characteristics can change over time due to change in land and water use patterns. Therefore, the watershed management framework should be adaptable to change.

It should be noted that watershed boundaries do not correspond to jurisdictional or political boundaries. An integrated watershed management approach is required to address the economic interests of stakeholders within different political boundaries and attempt to reconcile conflicting interests (USEPA 2007). An integrated management approach will incorporate all the initiatives and actions of regulatory agencies as well as local watershed associations. As noted in Chap. 2, the process of integrated watershed management is continuous (USEPA n.d.).

6.2.2 Source Water Pollution Control Measures

For effective source water pollution control, there is a need for effective legislative actions to control sources of water pollution (point and nonpoint sources) in target watersheds. A brief discussion of controlling point and nonpoint sources of pollution, and a brief description of ecological risk assessment (ERA) and its application to watershed management are provided below. ERA is practiced in the US and is worthy of serious consideration for other large areas where there is a risk of nonpoint source pollution.

6.2.2.1 Point Source Pollution

Point source pollution can originate from sewage treatment plants, industrial plant effluents, and animal farms. Point sources of water pollution are still a major problem in most developing countries due to lack of infrastructure, regulation, or its enforcement. In the U.S. and most other developed countries, the quality of effluents discharged from sewage treatment plants and industrial facilities is highly regulated, and thus these effluents do not generally pose a significant threat to the quality of receiving surface waters, unless wastewater treatment is inadequate or faulty. But animal production and farms are an exception. Below is a discussion of farm animal production problems and water pollution control in the United States, where animal production farms still pose a major threat to water quality. In the U. S., there are about 450,000 farms with animal feeding operations. About 85 percent of these facilities are small with fewer than 250 animals, but there are many animal feeding operations with more than 1,000 animals (USEPA 2002). These large farms are called "Concentrated Animal Feeding Operations" or CFAOs.

In 1998, the U.S. government released the Clean Water Action Plan. The Plan includes a "Unified National Strategy" for animal feeding operations, which tackles the large amount of manure and other wastes discharged by animal farms to water bodies. For CFAOs, farm owners/operators are required to have a permit that ensures safe disposal of all the manure, urine, and dead animal matter. The farms are subject to inspection and must have a comprehensive nutrient management plan that considers the safety of all nearby water bodies including groundwater. All CFAOs are required to keep records of the quantity of manure produced and how the manure was utilized, applied to land, sold to third parties for the manufacture of fertilizers, or used for methane generation as an energy source.

Apart from the regulatory requirements, there are voluntary guidelines from the U.S. Department of Agriculture (USDA) for best management practices (BMPs) on farms as well as tax incentives for demonstrating the implementation of BMPs. There are financial and technical assistance programs for implementing nutrient management plans as well as environmental education programs. And there are performance measures for the implementation of the "Unified National Animal Feeding Operations Strategy" (USEPA 2002).

6.2.2.2 Nonpoint Source Pollution

Nonpoint sources of pollution, also called diffused pollution, mostly originate from unknown origins and locations; it is the pollution that shows up downstream; it may include pollution due to the death of wild animals in water courses at unknown locations, or even bird feces, as noted in Chap. 2. Nonpoint sources of pollution associated with surface runoff include sediments, nutrients, pesticides, pathogens, metals, oils, and many chemical contaminants entering water bodies from unknown locations. Controlling nonpoint sources of pollution is rather difficult and complicated because of its diffused characteristics and difficulty in pinpointing the origin of contaminants flowing to surface waters. Watershed management and implementing BMPs are considered effective tools for nonpoint source pollution control.

6.2.2.3 Ecological Risk Assessment

To deal with nonpoint source pollution, the U.S. EPA recommends the application of ERA to watershed management (USEPA 1998). As defined by the U.S. EPA, "ERA is a process to collect, organize, and analyze scientific information in order to evaluate the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors" (USEPA 1998). Watershed ERA consists of a combination of one or more ERA methodologies and watershed management approaches. Table 6.1 shows major steps for a watershed ERA framework developed by the U.S. EPA (USEPA 2007).

Step 1: Problem formulation is defined as an integrated framework for risk assessment, including assessment of conceptual models and analysis plan. The conceptual models describe the various physical, chemical, and biological stressors, their sources, assessment endpoints and the possible pathways and also disclose how the assessment endpoints respond to the stressors via possible pathways, as shown in Fig. 6.1 (USEPA 2007).

Table 6.1 Steps involved in a watershed ecological risk assessment framework (USEPA 2007)

Step 1: Problem formulation
Translate proposed use into a goal and objectives
Develop conceptual model
Identify assessment endpoints and measures of impacts
Step 2: Risk analysis
Evaluate stressors, pathways, and measures of impacts
Use reference conditions and watershed data to determine effects on assessment endpoints
Step 3: Risk characterization
Compare stressor levels and physicochemical regimes with minimum thresholds or criteria for assessment endpoints
Determine relevant causes of nonattainability; stressor identification evaluation
Step 4: Risk management
Identify control options if applicable
Consider feasibility of specific controls
Conduct stakeholder discussions and management option evaluation
Modify the plan if necessary

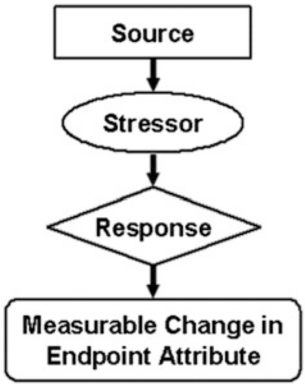

Fig. 6.1 Elementary conceptual model (USEPA 2007)

As an example, Table 6.2 shows the application of the assessment endpoints to respond to the stressors in the Waquoit Bay watershed.[1] A scoring approach is applied to evaluate the priority of stressors in the Waquoit Bay assessment. Professional judgment is used to rank the effects of stressors on identified assessment endpoints (Serveiss et al. 2004). Through the matrix given in Table 6.2, the risks associated with various stressors and their impacts can be estimated in a quantitative manner. Assessment endpoint can translate environmental management goals into a measurable system of attributes.

[1] See http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=162845 for a further description.

Table 6.2 Effects matrix summarizing assumed stressors and impacts on endpoints in the Waquoit Bay watershed (Serveiss et al. 2004)

Stressor	Assessment endpoints ranking						Totals
	Percent eelgrass	Finfish diversity	Scallop abundance	Anadromous fish	Wetland birds	Piping plovers	Fish/ shellfish	
Chemical pollution	1	1	1	1	1	1	3	9
Altered freshwater flow	1	1	1	2	3	1	1	10
Nutrient enrichment	5	5	5	3	2	1	1	22
Physical alteration of habitat	2	1	2	1	2	3	1	12
Fishing pressure	1	1	2	3	1	1	1	10
Pathogens	2	1	1	1	1	1	3	10
Totals	12	10	12	11	10	8	10	73

Each cell represents the relative effect of a stressor on an endpoint. The ranking (1 = minor, 5 = severe) reflects experience with the likely effects specifically for the Waquoit Bay watershed

In addition, the Risk Assessment Plan describes an approach to conducting the risk assessment. Note that the identification of stressors and their sources is essential to ensure the effectiveness of risk assessment. For example, the Tennessee Valley Task Force developed an inventory of environmental stressors to store information about nonpoint sources and point sources, and also developed solutions to solve the priority problems (USEPA 2007).

Step 2: The risk analysis phase searches for critical and influential stressors, forecasts how the stressors impact assessment endpoints via exposure pathways, and examines what human activities lead to changes in the ecological environment.

Step 3: The risk characterization represents the causes of uncertainties in both the problem formulation and analysis phases, and estimates the likelihood and the consequences of effects based on exposure and impacts. During the risk characterization phase, risk estimation is a challenge since natural resources respond to the multiple stressors through various pathways. Risk assessment should be evidence based. Furthermore, the quantitative information summarized and described by the risk description phase can be used to prioritize the estimated risks.

Step 4: Risk management refers to all the stakeholders making decisions based on the information made available by the previous steps. For example, they may rank and prioritize stressors and their consequent impacts. Past history may help in determining the degree of risk.

6.3 Risk Management Methods for Producing Potable Water Supplies

Major approaches for risk management to produce potable water discussed below include (a) the HACCP protocol, (b) WHO Water Safety Plan and the Bonn charter, and (c) Quantitative Microbial Risk Assessment (QMRA).

6.3.1 Hazard Analysis and Critical Control Point Protocol

The adoption of increased safety and reduced risk in drinking water has its origins in concern for food safety, and the realization that drinking water safety could and should be treated like food safety. Historically, in the Austro-Hungarian Empire (1897–1911), a collection of standards and product descriptions for a wide variety of foods was developed as the *Codex Alimentarius Austriacus*. This was a voluntary effort on the part of experts in the food industry and universities; the *Codex Alimentarius Austriacus* was not a legally enforceable set of food standards but was nevertheless used by the courts to determine standards for food safety (Davies 1970). It was to lend its name to the present-day international *Codex Alimentarius*

Commission, which is now under the administration of the Food and Agriculture Organization (FAO).

In the 1960s, the U.S. National Aeronautics and Space Administration (NASA) asked the Pillsbury Corporation to design and manufacture food for space flights. For safety, a protocol was devised to make sure that prepared foods were safe. This protocol became known as the HACCP protocol, which incorporated the systematic checks of the *Codex*. The HACCP protocol has since then received global acceptance as a procedure for handling and preparing food that is free of pathogens and is safe to eat.

The HACCP protocol is based on seven principles (Canadian Food Inspection Agency 2012):

Principle 1: Conduct a hazard analysis. Plans determine the food safety hazards and identify the preventative measures; the plan can apply to control these hazards. A food safety hazard is any biological, chemical, or physical property that may cause a food to be unsafe for human consumption.

Principle 2: Identify critical control points. A *critical control point* (CCP) is a point, step, or procedure in a food manufacturing process at which control can be applied, and as a result, a food safety hazard can be prevented, eliminated, or reduced to an acceptable level.

Principle 3: Establish critical limits for each critical control point. A critical limit is the maximum or minimum value to which a physical, biological, or chemical hazard must be controlled at CCP to prevent, eliminate, or reduce risk to an acceptable level.

Principle 4: Establish critical control point monitoring requirements. Monitoring activities are necessary to ensure that the process is under control at each critical control point.

Principle 5: Establish corrective actions. These are actions to be taken when monitoring indicates a deviation from an established critical limit. Corrective actions are intended to ensure that no product injurious to health or otherwise adulterated as a result of the deviation enters commerce.

Principle 6: Establish procedures for ensuring the HACCP system is working as intended. Validation ensures that the plants do what they were designed to do, that is, they are successful in ensuring the production of a safe product. *Verification* ensures the HACCP plan is working as intended.

Principle 7: Establish record-keeping procedures. The HACCP protocol requires that all plants maintain certain documents, including its hazard analysis and a written HACCP plan, and records the monitoring of critical control points, critical limits, verification activities, and the handling of processing deviations.

Any organization interested in risk minimization practice toward food and water can apply for certification for both the HACCP protocol and the International

Organization for Standards (ISO) protocol, ISO 9001.[2] The latter certification demonstrates that quality and customer satisfaction are priorities for the enterprise. The HACCP audits are conducted using auditor checklists based on *Codex Alimentarius* as well as local statutory and regulatory requirements. Food processors can be certified for ISO 9001 simultaneously, while an audit is conducted of their HACCP plans, resulting in certification for both. To provide food processors dual certification, it is possible to obtain a combined ISO/HACCP certification in preparation for the ISO 22000 standard for the food industry. ISO 22000 can be applied independently of other management system standards or integrated with existing management system requirements. The importance of ISO 22000 is that it integrates the principles of the HACCP system and application steps developed by the *Codex Alimentarius* Commission. Perhaps this is the standard to which water treatment plants will aspire in the future.

6.3.2 The World Health Organization Water Safety Plan

The use of HACCP for water safety was proposed by Havelaar (1994), and Iceland appears to be the first country to adopt the idea. The HACCP was also the basis for the Water Safety Plan (WSP) in the third edition of the World Health Organization (WHO) *Guidelines for Drinking-water Quality*, (2004), which has been described as "a way of adapting the HACCP approach to drinking water systems" (Rosén et al. 2008). The WHO HACCP-based framework, in particular the WSP, has been successfully applied to assessing and managing the risks posed by *Legionella* in building water systems (Bartram 2007). The WHO makes available both a Manual and an Excel-based management tool, which are available for use and can be downloaded from the WHO website (WHO 2004).

The template for a WHO-style Water Safety Plan emphasizes awareness of hazards of land use in general, and community education and outreach for water safety. This is particularly beneficial for developing countries where community education can do much to improve drinking water safety. There is also a simplified version for preparing a Water Safety Plan for Small Communities (WHO 2012).

Water Safety Plans have three major top-level components, which are then broken down into subcomponents. The three top-level components are:

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[2] International Organization for Standardization (ISO) is a worldwide network of national standards bodies from over 160 countries, which was established in 1947. The mission of ISO is to develop International Standards (i.e. ISO 9001, ISO 14000, ISO 27000, ISO 22000), and to make sure that goods, services, as well as processes are safe, reliable, and of good quality. As the management system standard, ISO 9001:2008 sets out the criteria for a quality management system implemented by over one million companies and organizations. To ensure that food is safe, ISO 22000:2005 contains the overall guidelines for food safety management, helping to identify and control food safety hazards. Detailed information can be found from http://www.iso.org/iso/home.html.

(a) A System Assessment: this component is a preliminary check to see if the drinking water supply chain as a whole is capable of supplying water of sufficiently high standard to meet regulatory targets of the country.
(b) Operational Monitoring: the higher order identification of the existence of control measures in the drinking water system.
(c) Management Plans: the documentation of system assessment and actions taken during various operational conditions; it also defines monitoring and communication plans.

The breakdown of these three components is described in Fig. 6.2. After setting up the Water Safety Plan implementation team and describing the supply system, the most important step is "conducting hazard analysis." The WSP team would consider all potential biological, physical, chemical, and radiological hazards that could be associated with the water supply. At each step, the objective is to identify where and what kind of contamination could happen and what are the set of actions that can be utilized to control each hazard. For example, the hazards may be due to variations in extreme rainfall events or major storms. The hazard may be due to accidental or deliberate acts of contamination at either the drinking water treatment plant or at the wastewater treatment plant that could compromise sanitation and hygiene.

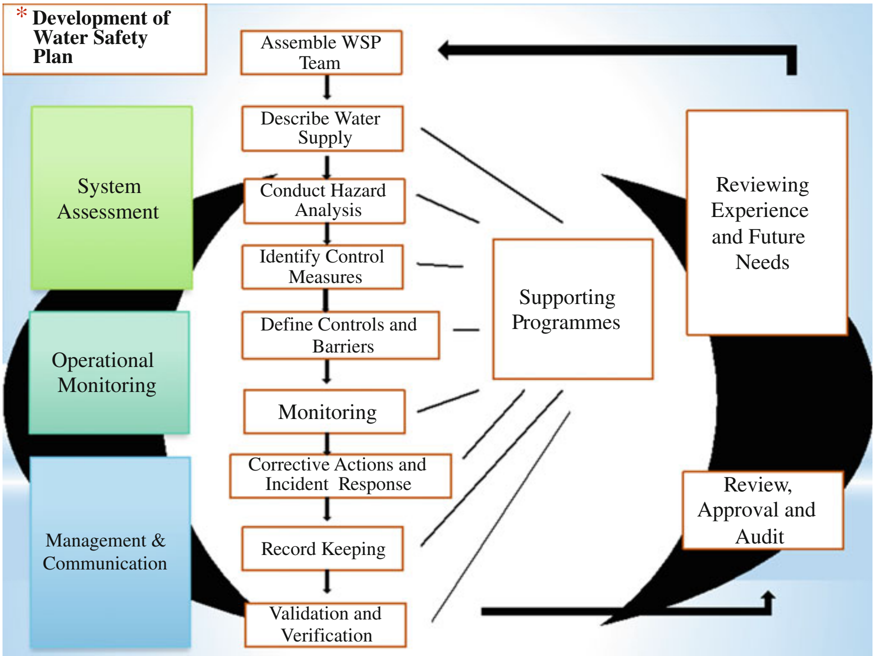

Fig. 6.2 Schematic overview of the WHO water safety plan components

Biological hazards include pathogens such as bacteria, viruses, protozoa, and helminthes. Other nonpathogenic organisms are asellus and Cyclops. These may originate from human or animal fecal material contaminating raw water that finds its way into the water supply delivery system.

There may also be radiological hazards as a result of contamination by man-made sources of radiation. These could arise from: naturally occurring radioactive species in drinking water sources; the contamination of water from the mining industry; or from radionuclides from the medical or industrial use of radioactive materials.

The risks are best prioritized by setting up a matrix of all relevant hazards. The team may rely on their a priori knowledge to assign "numbers" to the hazards or may weight hazards according to seriousness. These numbers will end up taking the form of subjective probabilities as objective probabilities will probably not be available.

Then "control measures" are defined as those steps that directly affect water quality, and which collectively ensure that water consistently meets health-based targets. They are activities and processes applied to prevent or minimize hazards. All actions that can mitigate risks from some specific events would also be documented. In particular, appropriate action is to be taken at the point of contamination, so that the effect of multiple barriers can be assessed together.

There are a number of ways to control pathogen entry into the water, for example, by reducing their entry into the water supply; reducing their concentration once in the supply; and reducing their proliferation. Another measure is source protection; decreasing contamination of source water will result in the amount of treatment and quantity of chemicals needed being reduced. This may further reduce the production of disinfection byproducts and reduce operational costs. Prohibiting polluting activities can enhance source protection. An important measure is promoting awareness in the community of the impact of human activities on water quality.

The next important step is "monitoring," which requires conducting a planned series of observations or measurements of operational and/or critical limits to assess whether the components of the water supply are operating properly. The monitoring should be applied to each control measure. It also requires establishing a relationship between control measure performance (determined by measureable parameters) and hazard control performance. For example, with or without a SCADA (supervisory control and data acquisition) system, it is still necessary to monitor chlorine residuals, pH, and turbidity, which will further indicate if more hazards are present or not.

A proper Monitoring Plan would include: a list of parameters to be monitored; sampling location and frequency; schedules for sampling; methods for quality assurance and validation of the sampling results; the proper interpretation of sampling results and any follow-up required; documentation and management of records, including how monitoring results will be recorded and stored. There will also be requirements for reporting and communication of results.

The Water Safety Plan is useless unless prompt and corrective action is possible when the results of monitoring indicate a deviation from an operational or critical limit. This is the point about *intervention.* When a water treatment plant is small and simple, the possibilities of intervention are severely limited. In that case, an elaborate Water Safety Plan may not be of much use.

We may define the functions of “supporting programs,” which are activities that ensure the operating environment, the equipment used, and the people themselves do not become an additional source of potential hazards to the drinking water supply. Supporting programs ensure good process control, good management, and good hygienic practices.

Verification and Validation require the use of methods, procedures, or tests in addition to those used in monitoring to determine if the water safety plan is in compliance with the stated objectives outlined in the water quality targets and/or whether the water safety plan needs modification and revalidation. This stage may also include review of monitoring control measures, microbiological and chemical testing, or review of the water safety plan overall so as to ensure that it is still in accordance with the original intent. In principle, a Water Safety Plan should be constantly reviewed and updated.

6.3.3 The Bonn Charter

While the WHO was developing its approach to risk minimization for drinking water, a group of water industry professionals first met in Bonn in 2001 and worked on establishing very similar but complementary principles. After their second meeting in 2004, the Bonn Charter was born; it is endorsed by the International Water Association (International Water Association 2004). While the Bonn advocates also endorse the WHO approach, the Bonn Charter principles are slightly different. They are:

(1) Management of the whole water cycle; (2) management control systems to be implemented to assess risks at all points throughout water supply systems and to manage such risks; (3) an integrated approach requiring close cooperation and partnership between all stakeholders including governments; (3) open transparent and honest communication between all stakeholders; (4) clear responsibilities of the different institutions contributing to the delivery of safe and reliable drinking water; (5) water that is safe, reliable, and esthetically acceptable; (6) the price of water should be set, so that it does not prevent consumers from obtaining water of sufficient quantity and quality to meet fundamental domestic needs; (7) any system for assuring drinking water quality should be based on the best available scientific evidence and be sufficiently flexible to take account of the different legal, institutional, cultural, and socioeconomic situations of different countries.

Thus, the Bonn Charter emphasizes consumer satisfaction and recognizes that drinking water is a public good and that good quality drinking water should not be priced out of the pockets of consumers. On the other hand, the WHO emphasizes health and safety.

6.3.4 Quantitative Microbial Risk Assessment

What follows is a brief overview of the QMRA model. The entire process of QMRA is displayed as a flow diagram in Fig. 6.3.

Step 1: To begin with, it is necessary to find the concentration of a particular pathogen as the number of microorganisms per liter of source water. This must be done by taking a large number of samples, which are required in order to be able to fit a probability density function (PDF). For each pathogen, there is a baseline

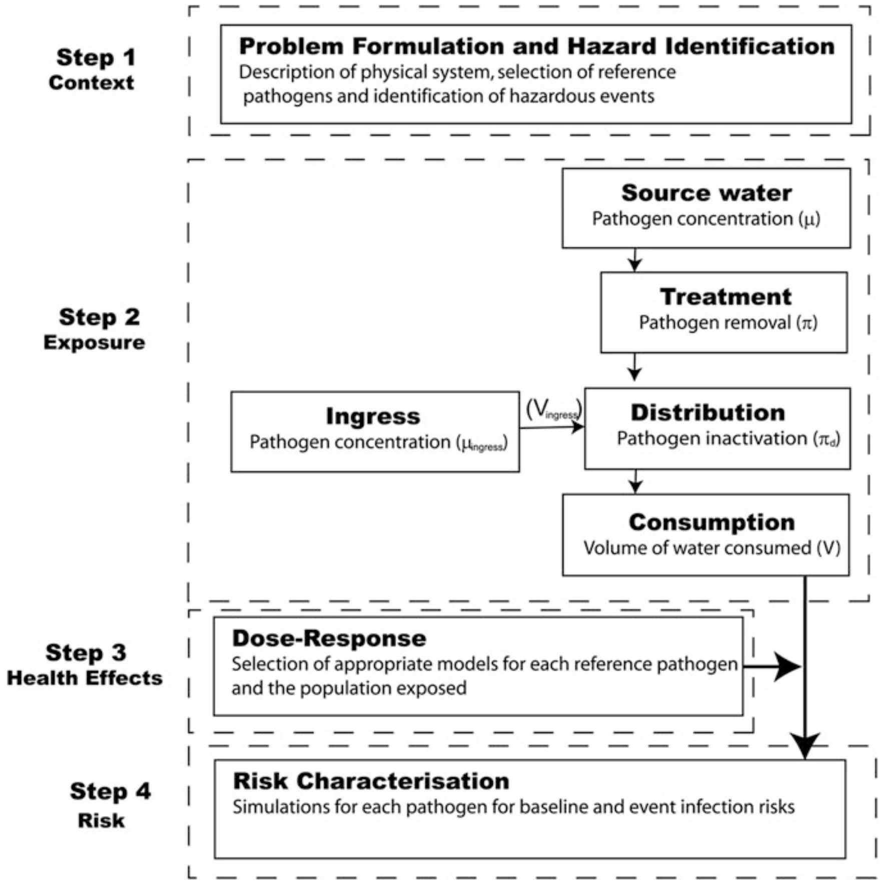

Fig. 6.3 General framework for calculating microbial risk from drinking water (Petterson et al. 2006)

concentration and an "event" concentration. (In this simplified coverage, the event concentration is ignored). Baseline concentration is given by the proportion μ, which is the mean probability density of a given pathogen.

Step 2 is the treatment stage. Here, we need a measure of how many microorganisms "pass" or evade the treatment barrier(s). Let the removal performance be represented as the proportion π. If we multiply π by the measure of concentration, μ, then we get a measure of the density of pathogens in the treated water entering the distribution network of pipes. But deficiencies in the pipes might lead to some addition of more pathogens. This additional intrusion of pathogens is sometimes due to *water-hammer* (unequal pressure inside and outside the pipe), due to some imperfections in the pipes, or simply due to the old age of the pipe. Figure 6.4 refers to this intrusion into the pipes ($\mu_{ingress}$) simply as "ingress"

The next step is water consumption. The "dose of pathogens" is the volume of water consumed, multiplied by the total pathogen concentration. This gives the "dose–response." After that we need to estimate the "probability of infection," which depends on the mean pathogen density and the number of organisms, n. Assume that the mean density can be represented by the Poisson distribution.

When infected water is consumed, the body's defenses might stop some pathogens. So, we now need the probability of the number of organisms that successfully overcome the defenses. This should be a Binomial distribution: you are either infected or not infected.

Next, the probability of infection depends on the probability of exposure to the number of infections and the number of organisms, given the pathogen removal, μ.

Next, we need an equation predicting the number of infections from multiple exposures. So far our prediction for the number of infections is a daily prediction.

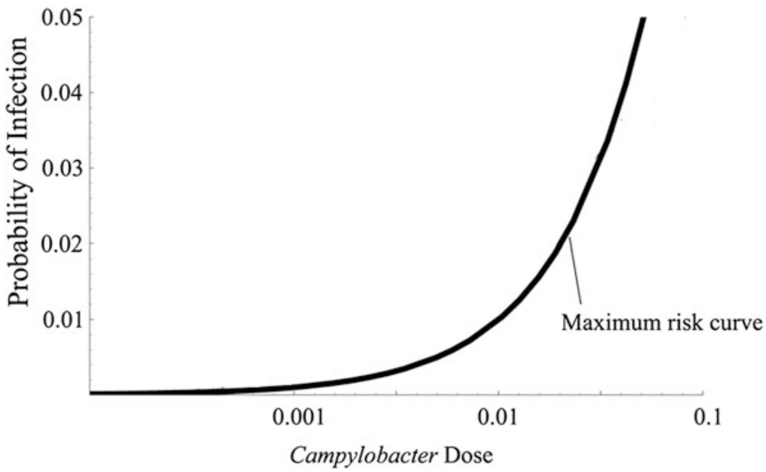

Fig. 6.4 Campylobacter and maximum risk dose–response curves at low doses (Petterson et al. 2006)

It can be shown that the combined probability of exposure and infection can be stated as

$$P(\inf|\,\mu) = \sum_{n=0}^{\infty} P(n|\mu) X P(\inf|\,n) \tag{6.1}$$

where

$P(\inf|\mu)$ is the probability of infection given the mean pathogen density

$P(n|\mu)$ is the probability of exposure to n organisms, given the mean pathogen density μ;

$P(\inf\ |\mathrm{n})$ is probability of infection given exposure to n organisms

We need one more probability: the probability of the organism successfully overcoming host barriers. Let that probability be r. If the organisms are randomly distributed, the probability of infection $P_{\inf}$ is

$$P_{\inf} = 1 - \exp(-r\mu) \tag{6.2}$$

assuming the randomly distributed organisms can be represented as a Poisson distribution.

A property of this type of model is that it can now be shown that a *maximum risk curve* exists and it takes the shape shown in Fig. 6.4, drawn for the pathogen Campylobacter.

6.3.4.1 Risk Characterization

The objective of risk characterization is to integrate information from exposure and dose–response models to express public health outcomes, which requires predicting the number of infections from multiple exposures. The number of infections may be described as a binomial random variable X. The probability that the number of infections will equal a given number k is given as

$$P(X = k) = \sum_{k=0}^{n} \binom{n}{k} p^k (1-p)^{n-k} \tag{6.3}$$

where k is the number of infections

n is the number of infections per year for an individual; $n = 365$ for the number of infections per year

p is the probability of infection.

This equation can be maximized to find the most likely infections based on the calculation of $P_{\inf}$ given in Eq. 6.2.

If we assume that the consecutive exposures are independent of each other, we can find the annual probability of one or more infections by assuming a binomial process. If the probability of infection is $P_{\inf}$, then the probability of NOT being infected is $(1 - P_{inf})$. For an annual probability, $n = 365$. So, the annual probability is given as

$$P_{ann} = 1 - (1 - P_{inf})^{365} \tag{6.4}$$

Now, infections are necessary to cause disease, but not all infections will result in symptoms of illness. In general, economists and epidemiologists express the disease burden in terms of *Disability Adjusted Life Years* or DALYs. We interpret DALYs to be years lived with a disability plus lives lost due to a hazard, such as an *E. Coli* outbreak. DALYs take into account both illness outcomes as well as the duration and severity of the illness. Thus, the calculation of DALYs per infection is given as

$$\mathrm{DALY} = \sum_{i=1}^{n} P(\mathrm{ill}|\mathrm{inf}) X\, P(\mathrm{outcome}_i|\mathrm{ill}) X\, \mathrm{Duration}_i\, X \mathrm{Severity}_i \tag{6.5}$$

where:

n	is the total number of outcomes;
$P(\mathrm{ill}\|\mathrm{inf})$	is the probability of illness given the infection
$P(\mathrm{outcome}_i\|\mathrm{ill})$	is the probability of outcome i given illness
$\mathrm{Duration}_i$	is the duration in years of outcome i
$\mathrm{Severity}_i$	is the severity weighting for outcome i

The advantage of using DALYs is that it can take account of not only illnesses like enteric illnesses but also more serious disease outcomes such as the Guillain-Barre syndrome associated with *Campylobacter*. Disease burdens vary widely depending on locality. For example, the disease burden per 1,000 cases of rotovirus diarrhea is 480 DALYs in low-income areas, whereas in high-income areas, it is only 14 DALYs per 1,000 cases. Other examples of disease burden estimates for drinking water contaminants are reproduced in Table 6.3.

The disease burden based on DALYs would be calculated using the number of infections per year, i.e. maximizing Eq. 6.3 above for the population multiplied by the DALY contribution per infection.

It should be obvious that the calculations involved are not so simple. But for freshwater beaches in southern California, QMRA has been used to determine the safety or otherwise of a particular beach.

6.3.4.2 Implementing QMRA

Figure 6.5 shows the main steps in the implementation of a QMRA procedure, from the first step of estimating the mean probability density of a given pathogen, to estimating the costs of infections based on DALYs per person per year. It is also necessary to distinguish between baseline hazards as well the extra "burden" of

Table 6.3 Summary of disease burden estimates for different drinking water contaminants (Havelaar and Melse 2003)

	Disease burden per 1,000 cases		
	YLD	YLL	DALY[a]
Cryptosporidium parvum	1.34	0.13	1.47
Campylobacter spp.	3.2	1.4	4.6
STEC O157	13.8	40.9	54.7
Rotavirus			
High-income countries	2.0	12	14
Low-income countries	2.2	480	482
Hepatitis A virus			
High-income countries, 15–49 yr	5	250	255
Low-income countries	3	74	77

[a] DALY is the sum of "years of life lost due to death" (YLL) and "years of life lived with disability" (YLD) (WHO 2014)

exogenous hazardous events such as extreme rainfall events. When that burden is "excessive," corrective, or remedial action is required at the critical control points. One example of such an action is increased disinfection to counter the effects of the exogenous (nonnormal) hazardous event.

It should be clear that the implementation of QMRA is nontrivial. For drinking water treatment plants, the computational requirements for using QMRA would be huge and the payoff is debatable. For this reason, it may be better to turn to other risk minimization methods. We offer the following additional points by way of an assessment of QMRA.

6.3.4.3 Critique of QMRA

An assessment of QMRA can be summarized as follows:

1. As the empirical CDFs (cumulative distribution functions) are unknown, it has been necessary to impose arbitrary CDFs of a known distribution such as the Poisson distribution, the Beta and Binomial distributions.
2. It is applicable to treatment systems where the known pathogen removal (π) is of "low" log reduction.
3. If the removal of pathogens is 4-log or higher, then the focus would shift entirely to possible ingress or regrowth in the pipeline network.
4. The effectiveness of each process in removing pathogens is variable between (a) the same processes operated at different treatment plans and (b) over time at the same plant. Thus, calculating removal performance at any time would be extremely difficult.
5. Calculating treatment effectiveness depends on the following data:

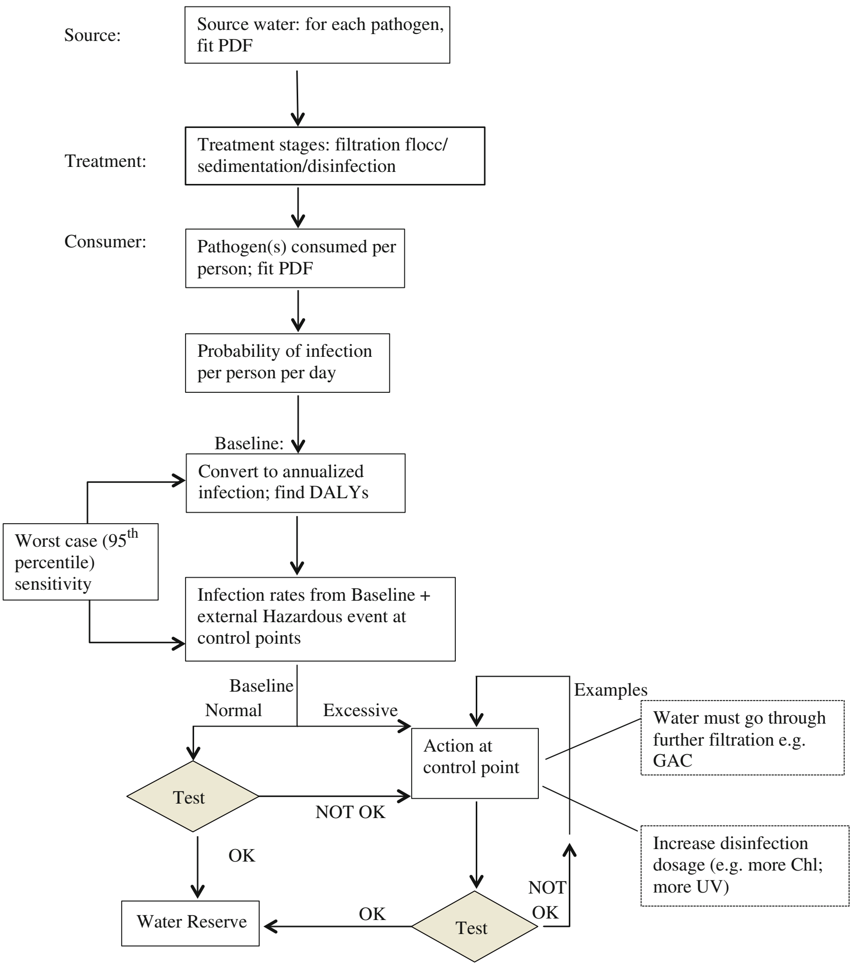

Fig. 6.5 Flowchart for the implementation of QMRA (Reproduced from Nilsson and Thorwaldsdotter 2006)

(a) Pathogen densities at intake and outlet of the treatment plant.
(b) If surrogate densities are used (in the absence of actual densities), this could add another layer of uncertainty.
(c) Reliability of plant SCADA systems for data on turbidity, chlorine residual, etc.

6. In the pipeline network, deficiencies may lead to (a) ingress or regrowth of pathogens, and (b) uneven flow which may dislodge biofilm pathogens from the interiors of pipes to enter the water column flowing through. These two events are not captured by QMRA.

7. In QMRA, the probability of infection, given a particular density of a pathogen, in turn depends on the probability of exposure to the organism. But the rate of infection depends on the individual's capacity of immune response. The only known "outcome" is "illness," but the illness intensity and duration are very variable.
8. The estimation of the probability of infection depends on "dose-response models." A critique of such models is outside the scope of this chapter.
9. Infection is a necessary condition for a disease, but not all infections result in symptoms of illness. This still leaves the problem of the severity and duration of illness.
10. The risk of disease burden is measured in DALYs. DALY is a good measurement as it is calculated as a product of the probability of each illness outcome with a severity factor and the duration in years. But what is an "acceptable" DALY is a political judgment. Many jurisdictions (for example, Canada and the US) accept the WHO recommended acceptable level of risk as 10^{-6} DALY/person per year.
11. The final outcome in the formula of DALY is a social cost of pathogens; it is a social opportunity cost. Instead of spending effort and money to implement QMRA, the same effort could be more effectively used to increase treatment effectiveness and thereby raise log reduction to 4 or 5. This can be done cheaply with the newer UV modules that can inactivate a large spectrum of pathogens. Thus, instead of using QMRA, incorporating an UV module in every treatment train would be more cost-effective. It will certainly reduce the mean concentration of pathogens in the treated water and thereby reduce the probability of infection and illness.
12. Smeets et al. (2010) show that an application of QMRA can be used to determine the efficiency of treatment under the WSP system. Their study (a) helps to determine the acceptable time of system failure during the treatment process by setting critical limits and taking corrective actions when necessary, for example, the 2.5-log reduction of pathogens requires that the system failure time is not more than 6 h per year; and (b) shows that high frequency of physical and microbial monitoring can improve treatment effectiveness, i.e. conducting the microbial monitoring every 15 min can increase the reduction of pathogens by 2 logs compared to daily monitoring (Smeets et al. 2010).Thus, the application of QMRA helps to reduce the likelihood of uncertainty in the management of microbial drinking water safety (Smeets et al. 2010). Nevertheless, the computation burden is enormous.

6.3.5 Risk Assessment Application to Water Treatment Plants

It is clear that risk assessment methods should be integrated into the entire water production process from source water protection to the delivery of potable water to the consumer. In this section, we consider the application of risk at water treatment plants. We relate risk to regulation and show that regulatory policy implies a measure of risk aversion, but that still leaves some residual risk. In addition, there must be some benchmark to establish the *acceptable* level of risk. As in other public policy domains, there is no such thing as being *completely* risk free.

Most water treatment processes in developed countries are required to consider risk, either implicitly or explicitly, in the regulations, such as 3-log reduction of cryptosporidium (99.9 percent reduction) and 4-log reduction of viruses (99.99 percent reduction). In addition to this regulation, the US now goes further in the direction of taking risk into account. For the US, under the 1996 Amendments to the Safe Drinking Water Act (§ 1458(a)(1), 42 U.S.C. § 300j-18(a)(1)), the U.S. EPA must consider susceptible subpopulations in its health risk assessments. The amendments mention specific groups, including young children, the elderly, pregnant women, and people who are immunocompromised by disease or treatment for diseases. The concept of susceptibility to adverse health outcomes from environmental exposures can be extended to other groups as well. But the statute expressly limits this undertaking to subpopulations that are "identified and characterized." In its first report to Congress on susceptible subpopulations under the Safe Drinking Water Act, in December 2000, EPA concluded that because "genetic influences are complex and still poorly understood," it "is unclear to what extent individuals with heightened sensitivities due to genetic factors meet the statutory criterion of 'subpopulations that can be identified and characterized'" (USEPA 2000). While directing the EPA to study and report to Congress its findings on susceptible subpopulations, the Safe Drinking Water Act is silent on whether and how EPA should apply these data on susceptible subpopulations in regulatory decisions for drinking water contaminants. However, the new amendments make it clear that MCLs for new contaminants are to be based on *risk assessment* and a cost–benefit analysis, i.e. the risk-based benefits must outweigh the costs of stipulating an MCL for a candidate contaminant.

It should be apparent that some degree of risk averseness is implied in the requirement of 3-log reduction of cryptosporidium and 4-log reduction of all viruses. But this is the "credit" granted to the water treatment plant; it does not by itself minimize the risk. The residual risk depends on the quality of source water; if the source water is very poor, the 3 and 4-log credits may not be sufficient to eliminate all risks of infection.

Of course the requirement of a multibarrier approach, i.e. the removal of pathogens at multiple stages between the source waters and the consumer, can be indicative of risk avoidance, but it may not be enough. Regulation reduces the probability of infection but it is not eliminated; depending on the quality of the source water, the treatment plant may need additional log credits.

Although all viruses should be considered, rotavirus is selected as the reference virus for risk assessment because of the prevalence of infection in children and the possibility of severe outcomes due to this virus. It is also assumed that if this reference virus were controlled, this would ensure control of all other similar viruses. If a source water has a mean concentration of approximately 1 rotavirus/100 L, a water treatment plant would need consistently to achieve at least a 4-log reduction in virus concentration in order to meet the acceptable reference risk level of 10^{-6} DALY/person per year. Thus, a minimum 4-log reduction and/or inactivation of viruses has become established as a health-based treatment goal. But many source waters may require more than a 4-log reduction as the treatment goal to meet the acceptable level of risk as explained next.

It should be noted that a source water concentration of 1 rotavirus/100 L of water generally represents groundwater sources and relatively pristine surface water sources. In North America, many surface water sources will have virus concentrations in the range of 1–100 viruses per 100 l of water or even more. In that case, obviously the treatment plant would need to strive for even higher log reductions in order to achieve the reference risk level of 10^{-6} DALY/person per year. Thus, if the source water concentration was 10 rotavirus per liter (1,000 per 100 L), then the treatment plant would need to have 7-log reduction in order to achieve the reference level of risk.

To summarize, (a) 4-log reduction of viruses is fine for good quality source water, such as groundwater or pristine water sources, but for most surface waters, the treatment plants would need to receive additional log reduction credits, possibly through a multibarrier approach, and (b) susceptibility to infection of subpopulations is still elusive: it looks as though science has not yet rendered an unambiguous definition of susceptibility and what is a "susceptible" population. But new candidate contaminants and their MCLs will be added only if the risk-adjusted benefits exceed the costs to the community. The cost–benefit analysis will reflect the acceptable risk as DALY/per person per year, specified as the community standard of acceptable level of risk. In other words, regulatory practice implies some measures of risk aversion. But since 1996, the US EPA goes further; it "builds in" risk assessment when a new contaminant is added with its MCL to the list of regulated substances. The "riskiness" of a contaminant is a deciding factor in whether the substance is controlled or not.

6.4 Case Studies of Risk Assessment

We present four case studies, two from developing countries, Bangladesh and Uganda; two from developed countries, Iceland and Australia. These case studies show the wide variety of application of risk assessment programs.

6.4.1 Bangladesh

As a developing country in South Asia, Bangladesh suffers high morbidity that comes from waterborne diseases such as diarrhea and dysentery due to poor sanitation and contaminated water. Furthermore, the drinking water in Bangladesh is principally contaminated by arsenic in the tube wells to levels above the Bangladesh standard of 50 µg/l, which is five times higher than the WHO Guideline Value (Arsenic Policy Support Unit 2006). Based on the third edition of the WHO Guidelines for Drinking Water Quality, Bangladesh began establishing and implementing Water Safety Plans (WSPs) for both rural and urban water supplies in 2004 (Arsenic Policy Support Unit 2006).

According to the WHO Guidelines, the health-based targets refer to a series of safety requirements and public health needs, which can be identified by four types: (a) health outcome (i.e. the reductions in risk of disease), (b) water quality (i.e. the concentrations of substances in water are considered to be of no risk to public health), (c) performance (i.e. the reductions in the concentrations of microbes in water through treatment processes), and (d) specified technology (i.e. the technology can meet the safety requirements) (Arsenic Policy Support Unit 2006). The Guidelines applied to Bangladesh contain three major elements, which are the following:

1. Water Safety Plans

In order to apply primary water supply technologies to the community in rural areas, “Model” WSPs were established through a consultative process promoted by the Arsenic Policy Support Unit (APSU) of Bangladesh (Arsenic Policy Support Unit 2006). With the development of the “Model” WSP, hazardous event analysis was conducted to (a) identify the threats, (b) assess the risks, and (c) determine the priorities for action. Subsequently, a series of Water Safety Plans was prepared for verification (i.e. Water Safety Plans for dug wells, pond sand filters, rainwater harvesting, and tube wells) (Arsenic Policy Support Unit 2006).

It is important to test the practical application of the “Model” WSPs and so a number of pilot projects were created by APSU. A diverse range of organizations carried out the pilots. These organizations were the NGO Forum for Drinking Water Supply and Sanitation; the Environment and Population Research Centre; and the Dhaka Community Hospital. The pilots were carried out in five districts of Bangladesh, namely Barisal, Dhaka, Chittagong, Rajshahi, and Sylhet (Arsenic Policy Support Unit 2006).

The aims of the pilot projects were (a) to observe whether all risks are identified, (b) to measure the performance of implementation of the “model” WSPs for different local conditions, and (c) to record the experiences and lessons obtained from the pilots. The overall performance of the pilot projects was positive and successful and the key findings can be summarized as follows (Arsenic Policy Support Unit 2006):

(1) Sanitary conditions and microbial water quality were significantly improved. Specifically, diarrhea morbidity was reduced by 12 percent in the final assessment compared to the baseline assessment.
(2) Application of the pilot projects in communities produced a positive feedback and WSPs were being widely accepted by communities. The community leaders directly improved the safety of the drinking water by using WSPs, particularly by moving sources of contaminants such as latrines and animal pens away from drinking water sources. They also cleaned the surrounding areas of the water supplies.

However, the performance of rainwater harvesters and shallow tube wells that are mainly provided by households were far from being successful. This may have been due to lack of training for household suppliers or due to other factors.

2. Surveillance

Surveillance refers to a process of water quality testing, sanitary inspection and audit undertaken to ensure the safety of drinking water supply, which was controlled by the Bangladesh Department of the Environment (Arsenic Policy Support Unit 2006). Although an independent agency would be more conducive to monitoring the performance of water supply, it proved to be difficult in rural areas. The Bangladesh Department of the Environment had limited capacity for implementing the surveillance program. For this reason, the Department of Public Health Engineering and some NGOs assisted in conducting the surveillance. However, the biggest challenge was the fact that the total amount of shallow tube wells had reached between 7.5 and 10 million in Bangladesh; most of the shallow tube wells were owned by households. Monitoring these was an impossible task.

Perhaps, an appropriate way of dealing with the challenge would be to focus on the community water supplies. Thus, the development of community monitoring tools is necessary in ensuring the quality of community water supplies. The community monitoring tools were used to conduct maintenance and to remove the sources of hazards in an appropriate manner, ensuring that there is a safe distance between sources of hazards and the water supply.

In practice, some village committees achieved good performance in the implementation of WSPs via community monitoring tools. However, some monitoring activities carried out by those responsible were not well documented. For example, as a part of the community monitoring tools, a record-keeping chart is used to record monitoring activities, but 58 percent of them did not complete the record-keeping chart (Arsenic Policy Support Unit 2006).

Removing the threats to health required adequate chlorination for dug wells and ponds; some households largely rejected the use of chlorine and very few households were found to continue chlorination over longer periods of time. It is unclear if the model WSPs made a real and long-lasting impact in Bangladesh; the 10 million tube wells do not appear to have benefited from the WSP. Finally, the rejection of chlorination by many households seems to indicate that after an initial decline in morbidity, it is likely to go back up again as the water will continue to be contaminated by pathogens. If such a basic action needed for water disinfection is not carried out, the merits of a Water Safety Plan will be seriously compromised. This experiment with WSP will require a major sustained effort including foreign aid if it is to succeed in Bangladesh. Thus, this experience with WSPs in Bangladesh can best be described as mixed.

6.4.2 Uganda

Uganda is a developing country in East Africa and adjoins Lake Victoria, the world's second largest freshwater lake. In 2008, more than 30 percent of the population accessed unprotected drinking water sources in Uganda (Gunnarsdóttir 2012). Protection of water sources is a big challenge, since the main water utility, Kampala Water, has no jurisdiction over the catchment of Lake Victoria, partly because the catchment of the lake lies in three countries (i.e. Uganda, Kenya, and Tanzania) and the legal framework of catchment protection between the three countries has not been established. In Kampala, the capital and largest city of Uganda, a number of problems associated with drinking water have appeared, for example, the quality of water has worsened and the application of filtration of drinking water to protect against cholera is expensive and not affordable (Gunnarsdóttir 2012).

The Water, Engineering and Development Centre (WEDC) in the UK as well as the National Water and Sewerage Corporation in Uganda (the two together are referred to as the "group") collaborated on a project (2002–2004) which is called "Risk Assessment and Management of Piped Urban Water Supply" (Tibatemwa et al. 2004). The aim of the project was to test whether the application of WSP to water supply would work in Kampala (Gunnarsdóttir 2012). The project developed the key steps, including (a) preliminary system assessment, (b) field assessment, and (c) Water Safety Plan.

1. Preliminary system assessment

The Kampala water system includes surface water, treatment, storage reservoir, and distribution. To assure its quality, a preliminary assessment of the system was carried out by the group using available data. By identifying the existing and potential vulnerability of the system and evaluating the likelihood of risk, the group found that contamination at a principal valve would likely cause greater risk than a valve in the tertiary infrastructure. Field assessment was introduced to define the different inspection points within the distribution system in Kampala and risks were identified based on the characteristics of each inspection such as population, pipes, and altitude.

2. Water Safety Plan

After the field assessment, a series of Water Safety Plans was developed for the Kampala system based on the five impact categories, which are the following (Godfrey et al. 2002): (a) mortality in large population; (b) mortality in small population; (c) morbidity in large population; (d) morbidity in small population; and (e) no detectable adverse effect.

The evaluation of the population impact was based on the degree of the threats and how many people would be impacted. Additionally, a function was created to evaluate the probability of risk and the extent of impact for each inspection point in the WSP. The project also contributed to monitoring and verification within the Kampala water supply system (Tibatemwa et al. 2004, p. 641). For each inspection point, preventative measures were conducted to assure that the targets would be achieved (Tibatemwa et al. 2004).

However, due to the fact that people outside the water utility of Kampala had no knowledge of WSP, the external audit failed to be productive (Gunnarsdóttir 2012). Moreover, the major barrier to implementing a WSP was a shortage of finance. This showed the need for a separate budget for the WSP. Furthermore, some inadequacies also existed in the process, which included (a) lack of involvement of the stakeholders and communities, (b) remaining risk for water quality, (c) high turnover rate of employees and inefficient training system, and (d) incomplete and poor documentation (Gunnarsdóttir 2012).

The general conclusion was that while a WSP approach can be applied in a developing country, in the case of Uganda, a more cost-effective approach was needed to develop risk assessment for water supplies (Godfrey et al. 2002). In 2008, an external audit of Kampala's WSP was carried out by the National Water and Sewage Corporation. Their results indicated that the distribution network system had expanded since implementation of the WSP; however, there did not seem to be any plan for continuous improvement built into the WSP. There was also no systematic documentation on incidents available for inspection when asked for, and there was no summary of incidents or deviation for each year. Finally, there was only a partly implemented training plan in place in Kampala. As a result, the staff were unable to carry out regular monitoring and bring valid results to the quality department (Gunnarsdóttir et al. 2012).

It seems that the potential benefits of a WSP were not realized due to lack of finance, lack of training, and inadequate record keeping. One is inevitably led to the conclusion that the preconditions for the successful application of a WSP did not exist in Uganda. This may be a lesson for other developing countries: the simple fact that there is a WSP does not ensure water will be free of pathogens.

6.4.3 Iceland

As one of the countries with the largest freshwater resources and highest quality groundwater in the world, Iceland's drinking water has been classified in legislation as a food to ensure the safety of drinking water since 1995 (Gunnarsdóttir 2012). Subsequently, in 1996, the Association of Icelandic Waterworks, Samorka, promoted the implementation of Water Safety Plans and created guidelines using the principles of HACCP (Gunnarsdóttir 2012). With the guidelines of the HACCP established, Icelandic Waterworks began applying it as a preventative approach for water safety management in 1997. There are many towns in Iceland that implemented this system including: Reykjavik and Vestmannaeyjar (in May 1997), Akranes (in April 2003) and Borgarnes (in 2004). At the critical control points for the towns, minor corrective actions and additional control measures were applied to drinking water management (Gunnarsdóttir and Gissurarson 2008). Since Icelandic Waterworks' initial attempt to apply the HACCP was too complex in structure, Samorka developed a simpler approach in cooperation with four small waterworks in 2004, which they called the five-step mini HACCP plan (Gunnarsdóttir 2012). The five-step mini HACCP included all the critical elements such as risk assessment, procedures for maintenance, control at critical points and response to deviations (Gunnarsdóttir 2012). The initial implementation of the mini HACCP program was a success for Icelandic waterworks, specifically through an increased awareness of the importance of protecting water resources, and also a number of corrective actions (Gunnarsdóttir and Gissurarson 2008). However, the first audit conducted on Iceland's implementation of the mini HACCP in some towns revealed a lack of an external audit, and inadequate internal self-regulation and control (Gunnarsdóttir and Gissurarson 2008).

In order to assess the performance of water utilities as well as analyze the correlation between different factors, a WSP(Water Safety Plan) scoring system was developed in 2009 (Gunnarsdóttir et al. 2012). The WSP scoring system was divided into four categories of performance, each with five items, and thus had 20 items in total. In particular, the categories were based on the principles of the well-known Plan-Do-Check-Act (PDCA) cycle that expresses the continuous improvement process in quality management. The four categories were as follows (Gunnarsdóttir et al. 2012):

- Category 1: Assesses the mapping of the hazards (Plan)
- Category 2: Assesses what actions were implemented (Do)
- Category 3: Assesses the documentation (Check)
- Category 4: Assesses the support actions that are used to maintain and improve the WSP (Act)

The WSP scoring system shows that reevaluation of daily execution and documentation was needed, especially with regard to audits which are crucial in maintaining and motivating continuous improvement of the WSP system at each water utility (Gunnarsdóttir 2012). In the assessment results from 16 water utilities in Iceland, most water utilities did well in mapping, risk assessment, and performance of the required actions to deal with any obstacles. However, there was a lack of documentation and supportive action such as external and internal audits and inadequate communication with the public (Gunnarsdóttir 2012). Nevertheless, it seems clear that the application of the mini HACCP is in general a success story.

6.4.4 Australia

A discussion within the Australian water industry had taken place about a new risk-based approach for managing drinking water quality. In 1996, the National Health and Medical Research Council (NHMRC) published the "Australian Drinking Water Guidelines." This document served as an addition to the guidance and standards associated with drinking water quality, and it was similar in approach to the WHO guidelines. Moreover, the document endorsed for the first time the use of quality management systems (McClellan 1998). However, in 1998, drinking water in Sydney was contaminated with Cryptosporidium and Giardia (McClellan 1998). This led to the adoption of a framework for managing drinking water quality that was incorporated into the "Australian Drinking Water Guidelines" in 2004 (Australian Government of National Health and Medical Research Council 2004). The Australian Government and the states and territories implemented a water safety plan called the "Framework for Management of Drinking Water Quality" (ADWG) and "Community Water Planner" (CWP), a tool for achieving quality drinking water in Australia (Byleveld et al. 2008).

1. Framework for Management of Drinking Water Quality in Australia

The framework played a critical role in drinking water quality management. It included elements of HACCP as well as ISO 9001. It provided information for all steps of drinking water production from source to tap and was used by the Australian community and the water supply industry (Australian Government of National Health and Medical Research Council 2004). The framework consisted of four general areas with 12 elements (see Table 6.4). The four areas are:

a. *Commitment.* To commit to establish a drinking water quality policy, regulatory and formal requirements.
b. *System analysis and management.* To assess the risk within the water supply system by conducting water supply system analysis and identifying the existing potential hazards and their sources as well as to measure proactively and control the drinking water quality.
c. *Supporting requirements.* To support management of the supply system through employee training, community involvement, as well as systematic documentation.
d. *Review.* To evaluate long-term performance, audit the effectiveness of the drinking water quality management system, and develop the plan to improve water quality management.

The framework was used to ensure that all the necessary elements of drinking water quality management were addressed and go beyond ISO 9001 and HACCP (Australian Government of National Health and Medical Research Council 2004). For example, as indicated in Table 6.4, the Australian framework and HACCP provide the identification of hazardous events as well as a systematic approach to

Table 6.4 Comparison of features from various management frameworks (Australian Government of National Health and Medical Research Council 2004)

Framework for Management of Drinking Water Quality in Australia	HACCP	ISO 9001
Commitment to drinking water quality management		
Element 1 commitment to drinking water quality management		
Drinking water quality policy		
Regulatory and formal requirements	c	c
Engaging stakeholders		c
System analysis and management		
Element 2 assessment of the drinking water supply system		
Water supply system analysis	c	
Assessment of water quality data		
Hazard identification and risk assessment	c	
Element 3 preventative measures for drinking water quality management		
Preventative measures and multiple barriers	c	a
Critical control points	c	
Element 4 operational procedures and process control		
Operational procedures	a	c
Operational monitoring	c	c
Corrective action	c	c
Equipment capability and maintenance	a	c
Materials and chemicals	a	c
Element 5 verification of drinking water quality		
Drinking water quality monitoring	c	c
Consumer satisfaction		c
Short-term evaluation of results		c
Corrective action	c	c
Element 6 management of incidents and emergencies		
Communication		
Incident and emergency response protocols		
Supporting requirements		
Element 7 Employee awareness and training		
Employee awareness and involvement		c
Employee training	c	c
Element 8 community involvement and awareness		
Community consultation		c
Communication	a	a

(continued)

Table 6.4 (continued)

Framework for Management of Drinking Water Quality in Australia	HACCP	ISO 9001
Element 9 research and development		
Investigative studies and research monitoring		
Validation of processes	[c]	[c]
Design of equipment		[c]
Element 10 documentation and reporting		
Management of documentation and records	[c]	[c]
Reporting		
Review		
Element 11 evaluation and audit		
Long-term evaluation of results		[a]
Audit of drinking water quality management	[c]	[c]
Element 12 review and continual improvement		
Review by senior executive	[c]	[c]
Drinking water quality management improvement plan		[c]

Notes
[a] Aspect not explicitly stated but interpreted as being included
[c] Aspect explicitly stated

assessing the risks for drinking water (Australian Government of National Health and Medical Research Council 2004). In comparison with HACCP, the framework is able to evaluate the potential risks in the drinking water system using qualitative and quantitative data (Australian Government of National Health and Medical Research Council 2004). However, as a fundamental quality management system, ISO 9001 does not include the identification of hazardous events in water supply systems, nor does it include an assessment of water quality data (Australian Government of National Health and Medical Research Council 2004).

2. CWP tool

In Australia, there are many remote and indigenous communities, in which the safety of drinking water has been a critical issue for decades due to poor sanitation and contaminated water. In 2005, the first version of the CWP designed as a microbiological-based tool was released by NHMRC to support the implementation of ADWG, which included information on preventing microbial, physical, chemical, and radiological risks in drinking water (Centre for Appropriate Technology 2012). The CWP was aimed at helping smaller water suppliers in remote areas identify the existing and potential hazards and their sources, assess the risk of the hazards and evaluate performance through a risk-based Water Management Plan (Byleveld et al. 2008). The Water Management Plan is based on microbiological factors, which cover (a) water quantity and quality, (b) water supply from source to tap, (c) infrastructure organization, and (d) management of incidents and emergencies (Byleveld et al. 2008). With the application of the CWP tool, the indigenous and remote communities were able to improve the management of both drinking water and wastewater (Byleveld et al. 2008). In 2011, the National Water Commission improved the tool and made it a web-based application (Australian Government of National Health and Medical Research Council 2011a). The tools are available for download free of charge for people who work with small communities on water supply management (Australian Government of National Health and Medical Research Council 2011b). Indigenous communities have adopted it and have conducted comprehensive trials and testing (Centre for Appropriate Technology 2012). Murray Radcliffe, the Acting General Manager of the National Water Commission pointed out "when the CWP was tried and implemented in 21 Indigenous communities, fewer people experienced water-borne diseases" (Australian Government of National Health and Medical Research Council 2011a).

3. The revisions of ADWG in 2004 and 2011

As the amount and purity of chemicals controlled by individual treatment of drinking water supplies during the manufacturing process are different, contaminants may be contained in the specific treatment chemicals (Australian Government of National Health and Medical Research Council 2004). In order to provide the treatment of drinking water supplies with guidance on drinking water treatment chemicals, in 2004, 34 fact sheets were designed to identify potential contaminants for the individual treatment chemicals (Australian Government of National Health and Medical Research Council 2004). Each fact sheet contained (a) the guideline value and its derivation, (b) a general description of the characteristics of the contaminants, (c) typical values in Australian drinking water, (d) methods for removing contaminants from drinking water, (e) measurement and detection techniques, and (f) health indicators (Australian Government of National Health and Medical Research Council 2004). The Australian government required that this information be reported in the Annual Drinking Water Quality Reports. In 2011, the revised ADWG contained 120 new physical/chemical fact sheets and nine new microbial indicator fact sheets (Australian Government of National Health and Medical Research Council 2011a). Another major improvement of the ADWG 2011 was the enhancement of operational monitoring, which assists the utilities in measuring drinking water safety under all conditions (Australian Government of National Health and Medical Research Council 2011c). Furthermore, the information sheets were developed to describe the common processes used to disinfect water such as chlorine, chloramines, and ultraviolet (UV) (Australian Government of National Health and Medical Research Council 2011c). As the final step in a water treatment plant, disinfection has to have the greatest impact on drinking water safety. Thus, it is important to develop specific guidelines for the disinfection process to reduce the potential for waterborne diseases (Australian Government of National Health and Medical Research Council 2011c).

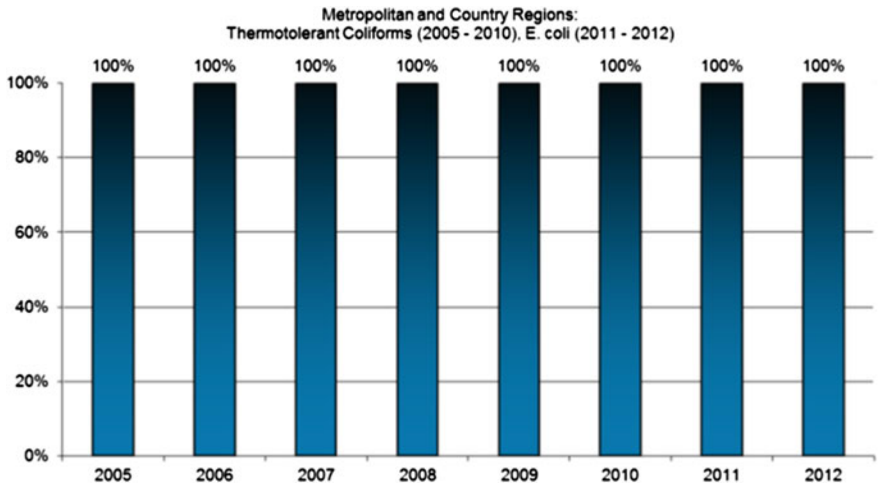

Fig. 6.6 Microbiological performance (2005–2012) (Water Corporation of Western Australia 2012)

i. Health-related performance

The Water Corporation of Western Australia, one of the largest water suppliers in Australia, provides drinking water to Perth and over 220 small communities throughout Western Australia (Water Corporation of Western Australia 2012). According to its Drinking Water Quality Annual Report 2011/2012, the microbiological performance target reached 100 percent in the metropolitan and country regions in Australia and remained at 100 percent throughout the period of 2005 to 2012 (see Fig. 6.6). In addition, all metropolitan and country regions achieved compliance with health-related chemical guidelines over this period (Water Corporation of Western Australia 2012).

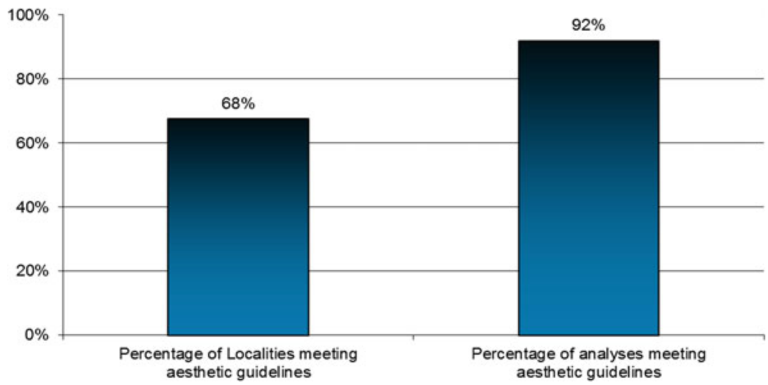

Fig. 6.7 Esthetic performance in Western Australia, 2011–2012 (Water Corporation of Western Australia 2012)

ii. Nonhealth or esthetic quality performance

The esthetic quality refers to the taste, color, and odor of drinking water. For example, iron and manganese residues within the distribution systems can cause water discoloration, which many consumers find unacceptable. Australian consumers were also dissatisfied with the use of chlorine as it leaves a metallic taste in the water. In 2011/12, around 68 percent of localities complied with all of the esthetic guidelines, while 92 percent of esthetic analyses for aluminum, color, and chloride met with the esthetic guidelines (Fig. 6.7). The remaining concerns could perhaps be attributed to (a) lack of alternative sources, (b) the impact of climate change on groundwater, and (c) groundwater abstraction near the coast (Water Corporation of Western Australia 2012).

It is a challenge to achieve the esthetic requirements of the Guidelines when there is limited availability of sources for drinking water and the cost of treatment is high (Water Corporation of Western Australia 2012).

6.5 Conclusions

The above review of the methods and tools of risk assessment for drinking water as well as the case studies show some interesting lessons. First, in the light of repeated outbreaks of waterborne disease even in developed countries, it is important to determine risk and decide on what level of risk public authorities are willing to accept, as there is no such thing as completely riskless water supply; there will always be unforeseen events and even accidents that can compromise water safety. Thus public policy should be guided by what are judged to be minimally acceptable risks in all sectors of public life. In drinking water, reducing pathogens by as much as 99.99 percent (4-log reduction) may be safe only if the source water is of sufficiently good quality. If the source water quality is poor to start with, then higher log reductions may be necessary. A multibarrier approach indicates that increasing source water protection can go a long way in reducing overall risks. In most countries, point source pollution control is the easiest to handle, provided there is political will and available financial resources. In many developing countries, point source pollution control either does not exist or the legislation is not being enforced adequately.

Second, nonpoint source pollution remains a significant threat to water supplies globally, and it is essential to establish some risk assessment framework, like that recommended by the U.S. EPA. For developing countries, the WHO-style "water safety plans" could work in principle, but as the case studies show, a precondition is the required education of the community and also an adequate administrative infrastructure to implement water safety plans. The WHO-style water safety plan *must* identify the *control point,* at which *the right intervention can be carried out.* Without the knowledge of the control point, there can be no *action* taken. Without the corrective action, water safety cannot be enhanced.

Third, our review above indicates that QMRA is in principle excellent but the computational burden of fitting PDFs to samples of water with pathogens is enormous. Even when that is done, it is necessary to determine the acceptable level of DALYs (DALYs/per person per year) that are politically tolerable and acceptable. This review argues that it is always (or almost always) cost-effective to add a newer UV disinfection module, rather than take all the mathematical and statistical steps in the numerical estimation of a QMRA model, because the new UV modules will definitely enhance safety by inactivating all pathogens. QMRA has its uses, such as determining beach safety for swimming, but for drinking water, faith in (cheap) *equipment* is a superior risk averse policy.

Fourth, the case studies indicate that the most successful risk minimization procedure is the mini HACCP protocol used in Iceland, and the similar planning tool utilized in Australia. Indeed, what can be simpler than treating drinking water simply as a food, as Iceland has done for a number of years? Both Iceland and Australia take risk very seriously and focus on the critical control points, where the required action can be taken. Australia has demonstrated that even remote indigenous communities can do this and thereby enhance their water safety.

The importance of risk assessment of drinking water is likely to become even more important, as the global impact of climate change will affect the quantity and quality of fresh water (Dore and Simcisko 2013). The new IPCC Fifth Assessment (AR5) (Intergovernmental Panel on Climate Change Working Group I 2013) confirms that patterns of precipitation are likely to continue to change; dry areas are likely to become even drier and wet areas are likely to get wetter (Dore 2005); and the frequency and intensity of hydrometeorological disasters is expected to increase. This will be seen in more extreme weather events, which will affect water turbidity and change the distributions of pathogens with increased variance. Earlier and faster spring snow melt will mean earlier runoff of water, causing more floods and also more forceful transport of natural organic matter in watercourses. Water treatment will no longer be "business as usual," but these weather events will force increased vigilance in water treatment and management and a greater reliance on risk assessment methods. We propose that all drinking water utilities aspire for certification under ISO 22000, which integrates HACCP with quality management.

Due to climate change, pathogens will continue to be a challenge. But once an acceptable threshold of pathogen-free water is secured, attention can then focus on the emerging chemical contaminants, such as pharmaceuticals and personal care products which consumers all over the world discharge into their wastewater that in turn end up in the source waters for drinking. This is the next frontier in drinking water treatment; this can be done, if there is political will, or if consumers demand this from their elected politicians. But for all countries, the first order of business is likely to be first monitoring and then effective elimination or inactivation of pathogens in drinking water.

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Chapter 7
Introduction to Water Infrastructure Asset Management

7.1 Introduction

A water utility typically has a large amount of physical assets made up of water treatment plants, pumping stations, reservoirs, and a network of distribution pipes. All these assets need to be used carefully and maintained for optimum performance. This requires an adequate inventory of all assets, including additions and scrapping and replacement of worn-out assets. This chapter reviews the key requirements of proper management of all these assets. In addition, we present a Decision Support System (DSS) that determines the number of years an asset must be used in order to minimize costs, when costs are considered over an infinite horizon. This DSS is presented in two forms: discrete and continuous time.

The second section of this chapter integrates the element of risk into the DSS in both discrete and continuous time. Risk is integrated in an abstract fashion in order to provide flexibility, as a municipality may decide to consider some risks and not others. It includes a brief discussion on the makeup of risk and the way in which different risk elements can be combined to introduce overall risk for a utility.

Great Britain was among the first countries to plan for municipal asset management. Privatization of water services in the 1980s resulted in the development of detailed asset management plans by these utilities. New Zealand, Australia, and the United States have also adopted accounting practices with respect to infrastructure in the late 1990s. Canada has also adopted modern accounting practices for infrastructure, including full cost recovery and depreciation, as opposed to charging all capital costs in the year in which they occurred.

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7.1.1 Infrastructure Management in Canada

Canada's Infraguide (2013), identifies 13 benefits of asset management including: facilitation and implementation of a plan; performance measurement; crisis aversion; cost and risk reduction; improved, continuous and consistent levels of service and better communication; return on investment and performance.

However, there are also three main challenges identified: asset management must be supported by senior management and comprise a key role in the municipality's business plan; the management plan must consider the life cycle of assets and the database utilized must be relevant and be up to date.

There are four key principles of asset management: it is "a strategic and proactive approach" (Infraguide 2013) that appreciates quality information and collaboration across both departments and disciplines; its vision is long term with respect to performance and cost with an emphasis on sustainability; it is a transparent method that obliges those involved to communicate in a meaningful way; and it involves business processes that base investment decisions that have clearly defined tradeoffs on policy and performance objectives.

The report identifies seven requirements of asset management. It must be recognized that all assets have a monetary value. There are seven stages in the life cycle of an asset (planning, design, construction, operation, maintenance, rehabilitation, and replacement); decisions in each stage have implications for other stages. The objectives must be sustainable in that the present needs are met without compromising the needs of the future. The asset management and financial plans should be integrated in order to enable the establishment of a quantifiable link between the level of service and the cost. Expected risk must be monitored (measuring all potential risks) to ensure that it is below some predetermined maximum tolerance level for risk. The performance of the assets must be monitored to ensure the ideal balance between cost, service, and risk.

The framework of asset management is described as a series of questions that address the policy objectives, funding limitations, inventory and condition, replacement costs, capital and operating plans and short and long-term financial plans. The policy objectives (level of service and acceptable cost) should reflect the values of the community and be directed by elected officials and municipal administration.

Asset management requires a detailed inventory (location, age, material, length and diameter of pipes, etc.). Replacement costs need to be enumerated to plan for infrastructure renewal. Life cycle costs and social costs must be considered for each asset. Estimated replacement dates for infrastructure need to be established and periodically updated; updates can be conducted through the assistance of deterioration modeling and predictions of usage. Based on the estimated replacement dates and renewal costs, projected cash flow requirements can be established to ensure that an adequate source of funds is available when needed; this will assist in cost containment and avert an unplanned service interruption. The financial plan should be of a temporal duration that would allow for possible cost increases; a duration that includes one life cycle of the longest lived component is sufficient.

The asset management plan depends on reliable and consistent data. The data must be collected in a standardized way, documenting links between different items, with a thorough database of the municipality's infrastructure inventory. Upon completion of the data collection, a commitment must be made to keep the database up to date; in some instances, a quality control plan will be necessary to ensure a continually improving quality of data.

Implementation of an asset management plan requires a business plan. The asset management team should include employees from all departments of the municipality. Asset information needs to be provided continually; this information should be shared amongst all team members.

As recently as 2002 the computerized systems available have been found to be inadequate as a sole source of asset management. These systems can only be considered to be a *part* of the plan; they cannot be considered to be the only plan. The implementation of the business plan should include projected benefits and costs. The implementation plan should include objectives, schedules, budgets, milestones, and responsibility.

The Technology Road Map was published by the Canadian Society for Civil Engineering, Canadian Council of Professional Engineers, Canadian Public Works Association, and the National Research Council (CSCE 2003). The road map includes both a survey of the current state of infrastructure in Canada and an identification of what will be needed in the future.

Benchmarking is very important in asset management as it provides a useful way of measuring performance. Metric benchmarking refers to the measurement of inputs and outputs against internal targets. Performance benchmarking refers to comparison with other entities that are known to perform well. Three benchmarking studies that were completed are the Ontario Municipal Chief Administrative Officers Benchmarking Initiative (OMBI), the Ontario Municipal Performance Measurement Program (MPMP) and the Canadian National Water and Wastewater Benchmarking Partnership. Any data collected from benchmarking must be considered in its environment. Different geopolitical conditions may mean that some entities are unable to meet the targets that others meet and can only use those targets as a guide.

Technology makes asset management easier; software enables utilities to collect, organize, manage, and present data in a number of different ways. Inspection and rehabilitation technologies allow the utilities to identify and repair weaknesses before costly accidents occur. The Ontario government passed the *Sustainable Water and Sewage Act* in 2002, which requires that municipalities record the full cost of water services and that they recover these costs.

Some municipalities have implemented programs designed to improve service delivery. Some of the programs that have been implemented are AWWA's QualServe Program, ISO 9000 and ISO 14000. ISO/TC 224 has been developed as the new standard for water (ISO 2014).

Asset management can be implemented in small and remote municipalities as well. Each plan needs to be adjusted to reflect the exact circumstances of the individual municipality. They should have maps that identify the type and age of each asset as well as its location and its identification number.

Small municipalities may not find it worthwhile to invest in a detailed asset management system or other expensive technology. However, they should still have a renewal plan for their assets. In Alberta the *Municipal Infrastructure Management System* (Alberta Transportation 2005) is available to assist small municipalities in tracking their inventory. The information can be used for forecasting, budgeting, and funding allocations.

An asset management plan needs to be monitored to ensure that service objectives are met over the long run. Monitoring can also ensure that the municipality is being managed efficiently. Technical reports can indicate the number of failures and be used to evaluate the performance of the system. Financial reports can be used to evaluate its efficiency. Unplanned spending should be tracked and should indicate a decrease upon implementation of an asset management plan. Total spending should also be tracked for each activity. The utilization of the benchmarks to compare the municipality to others can help determine areas of improvement.

7.1.2 Case Study 1, Capital Regional District of British Columbia

The Capital Regional District (CRD) of British Columbia provides water services for 13 communities on Vancouver Island. The CRD acts as both a wholesaler and retailer. Residents of smaller communities purchase their water directly from the CRD and all residents are metered. In larger cities the CRD sells the water to the city, which resells it to the residents. The CRD meters all water being sold to these cities. However, the cities do not always meter the water sold to residents.

The distribution network of pipes consists of 340 km of pipes up to 2.3 m in diameter. The primary water source has a capacity of 92.7 million m^3. The secondary source has a capacity of 10 million m^3. There are also two reservoirs for treated water that have a combined capacity of 50,000 m^3. Daily demand in the region varies from 114,000 to 318,000 m^3 of water. There are two treatment plants utilizing UV light and small amounts of chlorine. Residual treatment is accomplished by adding ammonia to the water.

Prior to the mid 1990s, asset replacement was primarily reactive, i.e. infrastructure items were only replaced after a breakage or failure. Their asset management plan was implemented in 1999 and the CRD now schedules renewal prior to failure. Its decision-making algorithm is displayed in Fig. 7.1.

This algorithm works through a series of questions to determine the appropriate life of an asset. Each asset is assigned a base life. The questions identify potential weaknesses in the asset. Upon identification of the weaknesses, the base life of the asset is adjusted to reflect the appropriate life.

The Asset Management Division is responsible for ensuring that aging assets are replaced prior to failure and that the infrastructure is appropriate for the district.

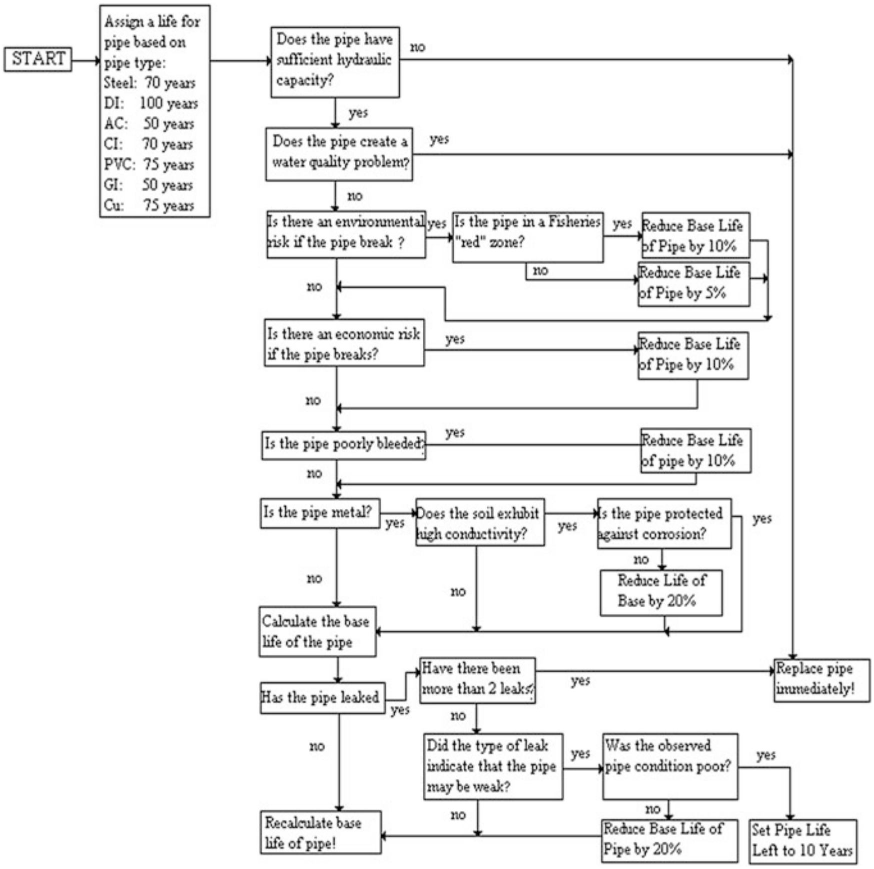

Fig. 7.1 The Decision-making algorithm used by the Capital Region District of British Columbia asset management

To accomplish this, they utilize maintenance and upgrade programs. Recent initiatives under these programs include the Main # 1 replacement (1994–2006) and the Sooke Reservoir expansion (1990–2002).

7.1.3 Case Study 2, Asset Management in Australia

The Queensland government of Australia has published an on-line[1] manual entitled Strategic Asset Management in order to assist its departments in the management of infrastructure assets. The manual was created in response to a “changing social,

[1] http://www.build.qld.gov.au/sam/sam_web/frames/guidelin.htm.

political and economic environment" (Queensland government of Australia 2002), in which government is under increasing pressure to provide better and more services while containing costs. It identifies a number of methods for managing infrastructure depending on the application. However, the basic concepts and foundation are consistent throughout the manual.

The objectives of the Australian approach are: structured and accountable corporate planning; establishment of a relationship between service delivery and resource planning; creation of plans for capital, maintenance and disposal; diffusion of appropriate processes to manage new assets; more effective and innovative service delivery; private sector participation in financing, provision, management, and maintenance of infrastructure; and enhanced coordination of public assets from a "whole-of-government" perspective.

In this approach, all infrastructure go through a 5-stage cycle: "plan, create or acquire, operate and maintain, refurbish or enhance, and dispose" (Queensland government of Australia 2002). The plan recognizes the fact that decisions at any one point in the life cycle have cost and output implications at other stages.

The asset management plan identifies the following six principles:

- Assets exist only to support the delivery of services.
- Asset planning is a key corporate activity that must be undertaken along with planning for human resources, information and finances.
- Non asset solutions, full life-cycle costs, risks and existing alternatives must be considered before investing in built assets.
- Responsibility for assets should reside with the agencies that control them.
- Strategic asset management within agencies must reflect the whole-of-government asset policy framework
- The full cost of providing, operating and maintaining assets should be reflected in agency budgets

Figure 7.2 shows the organization of the plan as a matrix for each stage of the life cycle. It has a 5-step approach to production: planning, investment, operational management, maintenance, and disposal of assets.

In meeting service demands, the utility must manage demand, maintain value and manage risk. Demand management is the process of moderating demand for a service. Value management involves finding the service delivery method that achieves the objectives at the lowest possible cost. Risk management entails identifying risk and methods by which to mitigate the size of the risk.

Traditionally, utilities have been directed from the top down. The instructions from higher levels were broad and became more detailed at lower levels. Unfortunately, as the instructions became more detailed the focus also became narrower. Appropriate asset management needs a "whole-of-government" perspective, and as plans become more detailed the focus must not become narrower.

Upon implementation of an asset management plan, a number of benefits should materialize. There should be a clear understanding of the purpose of the assets. Each asset should link to a specific service delivery objective. The capital should be in place to achieve the objective. Assets should be working properly and used in a

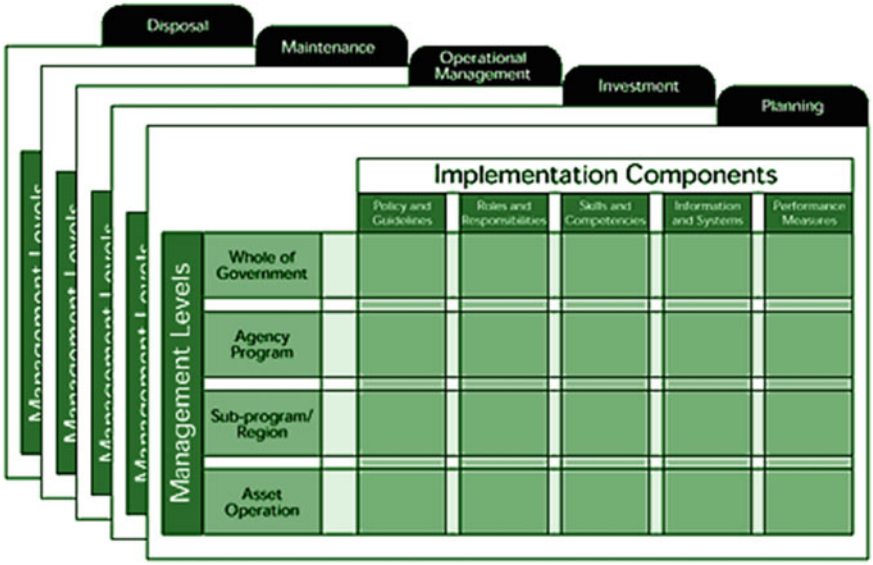

Fig. 7.2 The five matrices of the Australian strategic asset management

way that extracts the highest level of service from them. Economies of scale should be realized and the rate of return on the investment should be the highest possible. The plan should lead to appropriate environmental and workplace health and safety plans. Assets that are unused or not needed should be identified and decommissioned or liquidated. Information should be available as to the current value of assets at all times. Reserve funds should be utilized in a way that leads to optimum service. There should be an awareness of all opportunities and risks.

Since the objective of the utility is to serve the community, this service must be tracked. The best indicator of the performance of the utility and the asset management plan will be the level of service experienced by the community. Though individual assets and financial performance are important, the output level of service must remain the primary indicator of performance.

7.2 Incorporating Risk in Water Infrastructure Management

7.2.1 Introduction

This section introduces concepts that are important to the incorporation of risk into water asset management. The three types of failure that can occur in water distribution are discussed, as well as their inter-relationships. There are different types of costs that result from different types of failures. These costs and their relationship to the failures are discussed. Options for protection from risk in water asset management are identified. A case is made for risk reduction and management as opposed to risk removal or transfer. The concepts of risk tolerance and risk aversion are also discussed briefly.

It is also important to discuss the role of *redundancy* in a water distribution network. Three levels of redundancy can be considered: no redundancy, looped redundancy, and latticed redundancy. Each type is discussed in terms of its reliability. The section concludes with an explanation as to why latticed redundancy is superior to looped redundancy.

Completion of a risk assessment is an important step in the incorporation of risk into water asset management. All risks must be identified and measured in constant units. An acceptable level of risk must be determined. A plan must be implemented to ensure that the level of risk is acceptable. This plan must be continually monitored to ensure its effectiveness.

7.2.2 Risk Considerations

This section discusses some of the basic risk considerations that need to be understood before introducing risk into asset management. There are three primary types of failure. A Type 1 failure is best described as an equipment break and can lead to failure of Type 2 and 3, but cannot be caused by these failure types. A Type 2 failure is the intrusion of pathogens into the system and can lead to a Type 3 failure, but not a Type 1 failure. A Type 3 failure refers to service interruption, which can be a result of a Type 1 or 2 failure, but cannot cause another type of failure.

There are several costs of failure. The main costs fall into the categories of: repair of property; environmental and economic costs; the costs of temporary replacement; and the costs of communication and maintaining health. Each of these costs should be taken into account in relation to different failure types.

A standard measurement of risk is developed in the third part and includes both multiple risks and varying degrees of risk. Reliability of this risk measurement is also discussed.

The fourth part discusses different methods to protect against risk. Risk can be transferred to another entity; it can be avoided completely or it can be reduced and managed. In water distribution, transferring the risk by insurance or other means does not reduce the human cost. Complete removal of risk will render the service so expensive that most will no longer be able to afford this kind of water service. The optimal solution is to reduce and manage the risk.

The last part of this section discusses risk tolerance and risk aversion. An understanding of risk tolerance is necessary if risk is to be reduced and managed instead of transferred or removed. Risk aversion is important as it places a value on appropriate risk estimates.

7.2.2.1 Types of Failure

There are three failure categories associated with the provision of water: asset failure, pathogen intrusion, and service interruption.

Asset failure (Type 1) refers to a piece of equipment that ceases to function to at least some degree. This includes a break, seize-up or plug in any type of asset that renders the asset at least partially inoperable. A Type 1 failure may lead to failures of Type 2 or 3 and subsequently to mass illness or loss of life. Type 1 failures can be caused by natural disasters, aging or weak infrastructure or poor workmanship.

Pathogen intrusion (Type 2) refers to the infection of the water supply. This type of failure cannot lead to a Type 1 failure, but may lead to a Type 3 failure and/or mass illness. Type 2 failures are caused by system contamination, pressure reversals, natural disasters, poor workmanship in pipes or valves, or source contamination and Type 1 failures.

Service interruption (Type 3) refers to a level of service that is suboptimal. This includes complete or temporary cessation of service or reduced levels of service. Type 3 failures are caused by, and do not cause, Type 1 or 2 failures. Type 1 and 2 failures lead to Type 3 failures under two circumstances: suboptimal redundancy and multiple failures of Type 1 and 2. A Type 3 failure may also be a result of a combination of these two circumstances.

7.2.2.2 Types of Failure Costs

The costs of failure manifest themselves in a number of ways: repair, property damage, environmental or ecosystem impairment, economic losses due to failure of delivery, temporary replacement, communication costs, and health costs.

The most obvious and easiest cost of failure to enumerate is that of the impending repair. Any type of failure must be fixed. Type 1 failures necessitate the repair or replacement of the respective asset, disinfection, and cleaning. Repair costs associated with Type 2 failures refer to pathogen removal, disinfection, and flushing. Type 3 failures arise from Type 1 or 2 failures. The repair costs of Type 1 and 2 failures must be accounted for appropriately. Therefore, the repair cost of a Type 3 failure is simply the cost of repairing any damage to the system caused by the interruption (i.e. pumps running dry, flushing due to pressure reversals, etc.).

All failure types may result in environmental damage. Type 1 failures may result in flooding that may damage the local ecology. Any time that disinfection and flushing is undertaken and treatment administered, there are costs. In addition, infected water needs to be dealt with properly as improper disposal can be environmentally dangerous.

Economic costs need to be accounted for as well. Type 1 failures may result in repairs that necessitate road closures and traffic diversions. These inconveniences can disrupt a local economy. Some businesses will experience a loss of patronage; others may incur staffing problems, as employees take longer to get into work.

Shipping costs may also increase. Type 2 and 3 failures may result in illnesses leading to a greater number of sick days.

Temporary replacement costs refer to those costs that become necessary, as an alternative source of water needs to be mobilized. At times this cost may refer to an alternative for water as opposed to an alternative source. Typically, this will result from a Type 3 failure. However, this cost may be necessary under Type 1 and 2 failures as well. Communication costs refer to any costs originating from the need to communicate road closures, boil water advisories, etc. Health costs refer to all health costs resulting from illnesses caused by any of the failure types. These illnesses range from the very serious (say bacterial infection caused by *E. Coli*) to the very minor (slight dehydration). It is important that all costs are recorded only once. For example, health costs include all forms of medical treatment, but do not include lost wages, as lost wages are an economic cost. Hence, there should be no duplication of costs.

7.2.2.3 Risk Measurement

The primary measurement of risk is expected risk. This is simply the probability of a failure occurring multiplied by cost of that failure:

$$P_i C_i$$

where,

P_i is the probability that failure i occurs, and

C_i is the cost of failure i, should it occur

In the event that there are n possible failures that may occur, the expected cost of failure is the sum of all the probabilities of failure multiplied by their respective costs:

$$\sum_{i=1}^{n} P_i C_{ii}$$

where, P_i and C_i are as above, and n is the number of different failures.

In the event that a failure may experience varying degrees of severity, the expected cost of failure is the integral, from the lowest severity to the highest severity, of the probability of failure multiplied by the cost of the failure:

$$\int_{\underline{S}}^{\bar{S}} P(x)C(x)dx$$

where, $P(x)$ is the probability of severity x occurring, and $C(x)$ is the cost of severity x occurring.

S is the severity of the failure (the bar indicates the bounds of this failure)

$$\int_{\underline{S}}^{\bar{S}} P(x)dx = 1$$

The primary measure of risk can be supplemented by evaluating its variance. If the variance is low relative to the expected value of risk, then the expected value is a good measure and will be adequate much of the time. However, if the variance is relatively high, it indicates that the risk is volatile and that the actual cost of the risk may be very different from the expected cost. This is an important point, especially if risk aversion is considered.

7.2.2.4 Protection Against Risk

There are three ways to seek protection from risk: transfer, avoidance, and reduction. Risk transfer refers to the practice of transferring the risk to another entity. This can be accomplished through the use of insurance, outsourcing or other means. Actual risk does not change; rather, it experiences a lateral move from one entity to another.

Risk avoidance entails taking deliberate actions to eliminate risk. In the distribution of water, this may involve utilizing nuclear or space travel technology. Though these technologies may provide failure-proof water provision, the cost to render the service would be prohibitive. The concept of risk avoidance in water provision is contradictory in that the service is a necessity and avoidance is so costly that it is impossible to entertain risk avoidance completely as the cost becomes prohibitive. In effect, the only way to avoid risk completely in water provision is to stop providing water![2]

The water utility may also choose to engage in risk reduction. This involves the application of techniques and management principles to reduce the likelihood and consequences of risk. In water provision, this involves the optimal combination of maintenance and renewal that keeps risk at a minimal level. This third method is superior to the former two. The first method simply involves a lateral transfer. Some social costs will be too great to transfer via insurance. It is very difficult to put a price on an epidemic claiming the lives of thousands. Risk avoidance is not realistic in water provision. Nor can risk be eliminated completely. The best way to protect

[2] The long-term Boil Water Advisories that some jurisdictions issue is nothing short of avoiding responsibilities of a possible waterborne disease outbreak.

the population against the risks of water provision is to reduce the risk. The utilities need to identify a maximum level of risk and then maintain and renew assets in order to meet this standard in the most cost effective way.

7.2.2.5 Other Risk Considerations

There are other considerations that need to be addressed when discussing risk. Two of these other considerations that merit special mention are risk tolerance and risk aversion. Risk tolerance refers to a certain amount of risk being tolerable. If there is no risk tolerance, the only way to protect an entity against risk is to eliminate it—which has already been ruled out. A certain amount of risk must therefore be tolerable, and this amount must be determined.

Another consideration is that of risk aversion. Given the choice between two options that yield the same expected risk and varying degrees of variance of risk, the utility should pick the option with a smaller variance. The reason for this is that when operating within a narrower interval, the utility can reduce the chances of catastrophic risk. Moreover, if the risk turns out to cost a little more than the expected value, then the cost can be recovered through small price increases. If the cost turns out to be much higher, then these prices increases will have to be much larger. There is rarely opposition to utilities that wish to decrease their rates due to cost reduction, whether the cut is large or small. However, as utilities attempt to increase rates, the opposition will become much stronger as the increase gets larger. A smaller risk variance enables the utility to plan better and enjoy political support.

7.2.3 Redundancy

There are three essential levels of redundancy: no redundancy, looped redundancy, and latticed redundancy. Looped and latticed redundancies have the added benefit of no "dead ends." These "dead ends" lead to bacteria buildup due to lack of movement of water in the pipes. However, the focus of this section is to explore the different types of redundancy in order to determine which type will lead to the lowest probability of interruption. It will be shown that the best type is latticed redundancy, through which service continuity can be met without sacrificing health and safety objectives.

7.2.3.1 No Redundancy

A system without redundancy is also known as a straight-line system. The flow of water through this system originates from a reservoir or treatment plant and flows through a main of a given length. This main would consist of a number of connected pipes. Valves would also be spaced out throughout the system to create

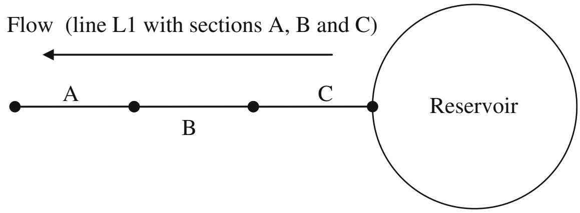

Fig. 7.3 Representation of a distribution with a single *line* of a pipe

"sections." The purpose of having sections is that the utility is able to stop the flow of water in or out of a section in the event of failure. However, since this system is a straight-line system, a failure at any point upstream will lead to a service interruption at all points downstream. Consider a given section of a main named A; the probability of service interruption at A is the union of probabilities of failure at A and all sections upstream from A. These upstream sections would be identified as B, C, D, etc. Since this is a straight-line system the sections A, B, C, D, … form a line. This line can be called $L1$ (see Fig. 7.3). Since the issue of concern is service interruption, this analysis will exclude failure in the section of concern, namely section A. The reason for this is that including the probability of the failure of section A would be redundant—no matter which level of redundancy is used, there is always this danger. Therefore, for simplicity and clarity, only an interruption of service to the section will be considered. Service interruption at section i is identified as I_i and failure at this section is identified as F_i. An interruption occurs at a given section when an interruption occurs at *any* section upstream. Alternatively an interruption occurs at a given section when a failure occurs at any section upstream.

Considering section A again: if the line is located on $L1$, which consists of 3 sections, A, B, and C, then the probability of interruption at A is $P(I_A) = P(F_B \cup F_C)$ or $P(I_A) = P(F_B) + P(F_C) - P(F_B \cap F_C)$; the overlap must be deducted since these events are not mutually exclusive. The probability of interruption on $L1$ leading to A is denoted as $P(L1) = P(I_A) = P(F_B) + P(F_C) - P(F_B \cap F_C) \leq 1$ (see Fig. 7.3).

7.2.3.2 Looped Redundancy

The above system can be extended to include a second line that would provide service to A. If this were to occur, the system would look like a loop (see Fig. 7.4). $L1$ would have the same probability of interruption as identified above and $L2$, consisting of the same material, would have an equal probability of interruption: $P(L1) = P(L2) \leq 1$. However, in order for service to A to be interrupted, both lines must become interrupted. Therefore, the probability of interruption to A can be expressed as the product of the probabilities of interruption on each line:

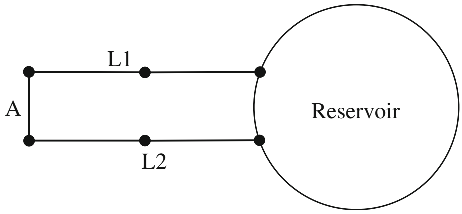

Fig. 7.4 Redundancy reduces the probability of failure

$P(I_A^*) = P(L1)^2$. Since $P(L1) \leq 1$, then $P(I_A^*) \leq P(L1)$—adding the loop has reduced the probability of interruption. Therefore, Looped Redundancy is better than No Redundancy.

7.2.3.3 Latticed Redundancy

Prior to introducing the lattice proper, the concept of a multiline loop should be discussed. Following from the above discussion of Looped Redundancy, we concluded that the probability of interruption decreased once another line was added. It is obvious that adding a third line would further decrease the probability of interruption. In fact multiline redundancy can be expressed in generalized form as $P(I_A^{**}) = P(L1)^n$ where n represents the number of lines servicing A (see Fig. 7.5).

In a multiline system, service to A becomes interrupted if all the lines become interrupted (experience a failure) regardless of the point at which this occurs. If the multiline system is expanded to resemble a lattice, then all the lines could experience failures, yet A may not lose service. The only way that A would experience a service interruption would be if all the lines experienced a failure at the same zone. Let $P(Z)^n \leq 1$ represent the probability of any given section experiencing a failure. In a latticed system with n lines of two sections each the probability of interruption to A is $P(I_A^{***}) = P(F_B^{***}) + P(F_C^{***}) - P(F_B^{***} \cap F_C^{***})$, where the three stars denote the latticed system. This probability resembles that of the single line system. However, in the single line system $P(F_i) = P(Z)$, and in the latticed system

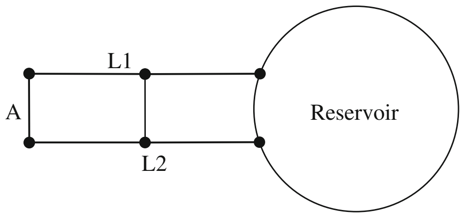

Fig. 7.5 Multiline redundancy reduces the probability of failure

$P(F_i^{***}) = P(Z)^n$. Since $P(F_i) \geq P(F_i^{***})$, clearly the probability of service interruption to A is lower under a latticed system than under a single line system.

Latticed Redundancy is more effective at reducing the probability of service interruption than both Looped and Multiline Redundancy. This is intuitive—the probability of interruption under a Multiline Redundancy includes that of the Latticed Redundancy and that of failures occurring on every line without any zone (see diagram) having n failures: $\mathrm{P}(I_A^{***}) = \mathrm{P}(I_A) + \mathrm{P}$ (every line experiences failure | no zone experiences n failures). It follows that $P(I_A^{***}) \geq P(I_A)$. That is, Latticed Redundancy is better than Multiline Redundancy. Therefore, by transitivity, Latticed Redundancy is better than Looped Redundancy.

7.3 Risk Assessment

In order to manage risk properly, a comprehensive risk assessment must be conducted. In developing an asset management plan, much information must be collected. It would be ideal to collect information pertaining to risk in this data collection stage. All risks must be identified. Many of the risks relating to water provision have been identified earlier in this chapter. Each of these must be identified in the risk assessment. Additionally, any risks unique to a specific utility must also be identified. Utilities may have risks specific to their geographic location, population size, water source, local ecology, and so on. Any special features must also be identified. From these features, any special risks must also be documented.

Upon documentation of all standard and special risks, the costs of these risks must be determined. When determining these costs, it is necessary to measure them in consistent units. Empirical evidence should indicate the cost and probability of the risk. For those risks with a stochastic element, the variance of the risk and probability distribution must be calculated.

The next step in the risk assessment is to determine an acceptable level of risk. In the water industry, the state or provincial level of administration or indeed a higher national level of government generally provides standards. However, in the event that these standards are not provided, the utility must determine the standard on its own. Upon determination of its water standards, the utility must place a value on the costs of not achieving these standards. This valuation will assist the utility in determining an acceptable level of risk. It will also assist in determining the value of risk reduction.

The final step in the risk analysis involves the monitoring of the risk management plan. Risk assessment needs to be continuous. As new risks are encountered, the assessment must be updated. The effectiveness of the risk management plan needs to be monitored as well. This will indicate whether or not the plan needs to be revised. Finally, the risky situations need to be constantly re-evaluated. Any anticipated changes in risk are best dealt with proactively and in anticipation rather than reactively.

7.4 Decision Support System (DSS) Incorporating Risk

7.4.1 Introduction

The occurrence of deferred maintenance in North America has left municipalities with outdated and weakened water infrastructure and the challenge is one of renewing this infrastructure in a cost-effective manner. In this chapter, we consider some key requirements for managing and renewing infrastructure. Specifically, we develop a mathematical-statistical model to determine the optimal renewal interval —the number of years an asset must be used before replacing it[3] through cost minimization. The DSS is the tool that is utilized to determine this interval. Subsequently risk is built into the model in an abstract fashion that allows the specification of a variety of risks.

7.4.2 The Decision Support System

Though Asset Management is a broad concept, one of the key objectives is to minimize the cost of a given service level by appropriately timing asset renewal. The time periods that utilities are concerned with are long term, but heavier emphasis is placed on the immediate short term. Therefore, present value discounting must be employed. The specification of a finite planning horizon is not necessary since the discount rate will ensure convergence and produce a meaningful solution. An infinite planning horizon actually allows for a simpler mathematical representation. Moreover, it is consistent with the fact that a public utility is expected to last for the indefinite future.

A DSS will factor in the service cost of a given asset discounted over an infinite planning horizon. The utility is faced with a decision in each time period and must decide whether to renew or maintain the asset. The solution to the DSS will identify the optimal length of service life and the point in time at which to cease maintenance and renew the asset. This solution will identify the optimal time to renew. In all prior time periods, the decision should be to maintain and use the existing assets.

The decision is based on the information input. At a minimum, the DSS must include investment and regular maintenance expenses over the life of the asset. However, other information can also be built into the model. Factors such as bedding conditions, volume, and quality of installation may necessitate increased maintenance, especially as the asset increases in age. These factors can be built into projected regular maintenance.

The model below rests on a few assumptions. It assumes that maintenance in period one, that is the time period of actual investment, is always included in the investment cost. Maintenance throughout the life of the asset can be represented as

[3] Based on present technology.

a function of the age of the asset, whereas the investment is a constant. The rationale for these assumptions is intuitive. Including maintenance in the investment cost does not change the model; it merely moves that period's maintenance into investment. However, the mathematical representation becomes less convoluted as a result—the model is simpler to understand. Representing maintenance cost as a function of age simply allows for more flexibility in the model as the expense may vary with age.

The DSS can be formulated in discrete and continuous time and the following sections are a mathematical statement of both.

7.4.2.1 The DSS in Discrete Time

In discrete time, the DSS can be described as a recursive function that includes the sum of the maintenance up to the period before replacement, the replacement cost itself, and future service cost, all discounted accordingly:

$$C_t = \sum_{i=1}^{t-1} \frac{M_i}{(1+d)^i} + \frac{I}{(1+d)^t} + \frac{C_t}{(1+d)^{t+1}} \tag{7.1}$$

where

M_i maintenance in period i

I investment

C_t aggregate cost of service over an infinite horizon for a given length between renewals (t)

d discount rate employed by utility for cost benefit analysis

Isolating C_t yields:

$$C_t = \frac{(1+d)^{t+1}}{(1+d)^{t+1}-1} \sum_{i=1}^{t-1} \frac{M_i}{(1+d)^i} + \frac{I(1+d)}{(1+d)^{t+1}-1} \tag{7.2}$$

The objective is to choose the time period, t^*, that minimizes C_t. This t^* can be found through dynamic programming or via a transformation to continuous time.

7.4.2.2 Dynamic Programming

The above objective can be solved using a deterministic Markov method; the utility must simulate the decisions that it will be faced with over an infinite time horizon. The state variable is:

$$t \in [0, \infty]$$

This is the age of the asset. The utility must decide to replace or continue to use the asset as a result of this information. The action variable is:

$$x \in [\text{keep, replace}]$$

These are the choices that the utility has. The state transition function is:

$$g(t,x) = \begin{cases} t+1, & x = \text{keep} \\ 1, & x = \text{replace} \end{cases}$$

If the utility decides to replace the asset, the age is reset to zero and next year's age is 1; if the utility decides to keep the asset, the age increases by 1 unit. The reward (present cost) function is

$$f(t,x) = \begin{cases} M_t, & x = \text{keep} \\ I, & x = \text{replace} \end{cases}$$

The reward will be either the maintenance cost or the investment cost. The value (present and present discounted future cost) of the asset is given by the Bellman equation:

$$V(t) = MIN\left\{M_t + \frac{V(t+1)}{(1+d)}, I + \frac{V(1)}{(1+d)}\right\}$$

The goal of the utility is to minimize $V(t)$. The utility must simulate the decision process at each and every future time period to determine when it will choose to replace the asset. The solution to this decision process will minimize the total present and future cost given in Eq. 5.1.

7.4.2.3 The DSS in Continuous Time

Equation 7.1 can be converted to continuous time as follows:

$$C(t) = \int_{1}^{t-1} M(x)e^{-xd}dx + Ie^{-td} + e^{-d(t+1)}C(t) \tag{7.3}$$

Isolating $C(t)$ yields:

$$C(t) = \frac{\int_{1}^{t-1} M(x)e^{-xd}dx + Ie^{-td}}{1 - e^{-d(t+1)}} \tag{7.4}$$

where:

t	is length of time the asset is used before it is renewed
$C(t)$	is the total cost over the infinite planning horizon as a function of t
$M(x)$	is the cost of maintenance as a function of age (x); and all other variables are the same as previously described

This objective can be solved via continuous optimization upon selection of the appropriate functional form for $M(x)$ and values for I and the parameter d. A generalized solution in terms of t is very difficult and nearly impossible to derive due to the existence of discounting. However, discounting is very important to this model as it ensures that measurement is consistent and that the solution converges. The paradox of this model is that without discounting no solution exists and with discounting the generalized solution cannot be isolated. The function $M(x)$ must be determined via regression analysis in order to solve Eq. 7.4 for the optimal t, which is t^*.

7.4.3 Incorporation of Risk into the DSS

This section outlines the process of incorporating risk into the basic DSS. In order to do so, a notion of risk must be established. An abstract characterization of this is used for purposes of risk incorporation: the functions S and F, as described below. The cost of failure, should it occur, is represented by S and F represents the frequency or probability of failure. The product of these two terms is the expected cost of risk. Where F represents a probability, $S \times F$ is the expected cost of failure at a specific point in time. Where F represents a frequency, SF is the expected cost of failure over an interval of time.

7.4.3.1 Discrete Model

To incorporate risk into the discrete DSS, first SF must be expressed in discrete terms—S_iF_i. Then, the expected risk cost in each time period must be added to Eq. 7.1:

$$C_t = \sum_{i=1}^{t-1} \frac{M_i + S_i F_i}{(1+d)^i} + \frac{I}{(1+d)^t} + \frac{C_t}{(1+d)^{t+1}} \tag{7.5}$$

The cost function still represents the aggregate flow of costs, discounted to the present value. However, this flow now distinguishes between regular maintenance, M, and expected accident costs, SF. Again, the goal is to minimize this eternal cost function. To do so, C must first be isolated:

$$C_t = \frac{(1+d)^{t+1}}{(1+d)^{t+1} - 1} \sum_{i=1}^{t-1} \frac{M_i + S_i F_i}{(1+d)^i} + \frac{I(1+d)}{(1+d)^{t+1} - 1} \tag{7.6}$$

The eternal flow, C, can now be minimized in two ways: via dynamic programming, as outlined in above, or through the transformation to continuous time that follows.

7.4.3.2 Continuous Model

In continuous terms SF becomes $S(x)F(x)$ allowing the transformation of Eq. 7.5 to continuous time as follows:

$$C(t) = \int_1^{t-1} (M(x) + S(x)F(x))e^{-xd} dx + Ie^{-td} + e^{-d(t+1)} C(t) \tag{7.7}$$

Isolating C yields:

$$C(t) = \frac{\int_1^{t-1} (M(x) + S(x)F(x))e^{-xd} dx + Ie^{-td}}{1 - e^{-d(t+1)}} \tag{7.8}$$

As in Sect. 7.4.3.1, it is desirable to minimize eternal cost with respect to t. For a practical application this can be accomplished through continuous optimization or numerical approximation. A numerical approximation may be a more reasonable approach considering the difficulty of isolating t and the 'lack' of precision required for practical applications. Asset management necessitates rounding of time at some point as it is not reasonable that utilities can measure time in infinitely small amounts. Nevertheless, if greater precision is desired, continuous optimization may provide a precise, meaningful and practical solution.

The above discussion on a solution to the DSS focused on practical solutions since, as identified in Sect. 7.4.2.3, a generalized solution to this problem is nearly impossible to derive.

7.4.3.3 What Are *S* and *F*?

In a nutshell, S and F are the modular components of risk. They can represent any type of risk that the utility is concerned with. The utility is limited only by the information it possesses. If data exist on the frequency and costs of a variety of accidents then these events can be statistically modeled and included in the DSS to determine the optimal renewal time in light of these risks. The utility can also simulate risk reductions and the impact that this may have on the renewal interval and on costs over the infinite horizon.

These have been referred to as modular components since they can be handled separately from the rest of the model. A number of different risks can be enumerated and included into this module that can, in turn, be used into the DSS. The utility will be faced with n different risks, each with its own risk cost in period i. The risk weighted cost of risk j in a given time period is:

$$S^j F^j$$

The aggregate risk weighted cost in a given time period is:

$$SF = \sum_{j=1}^{n} S^j F^j$$

where, in discrete time:

$$S^j = S_i^j$$

$$F^j = F_i^j$$

$$SF = S_i F_i$$

and, in continuous time:

$$S^j = S^j(x)$$

$$F^j = F^j(x)$$

$$SF = S(x)F(x)$$

Using statistical information in the specification of SF, this module can be included into the DSS to derive the optimal renewal interval.

7.5 Conclusion

The DSS would be used by a utility in conjunction with its existing 20-year strategic plans or life cycle plans. The solution to the DSS would indicate the optimal time at which to renew a given asset. The renewal cost would then be integrated into the utility's plan to ensure that funding is available when required.

Throughout this chapter, risk is handled as an abstract concept. In fact, all the elements of the DSS are handled in this way. The next chapter provides statistical methods for an application of the various components of the model. It also demonstrates the way in which an optimal solution may be determined.

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Chapter 8
Computing a Model for Asset Management with Risk

8.1 Introduction

In this chapter, the model presented in Chap. 7 is calibrated and solved with three case studies, which also illustrate the importance of using nonlinear methods. The decision support model would assist utilities in choosing the optimal renewal period for assets. The minimum of the cost function identifies the renewal period that results for the lowest expected service cost. This chapter discusses the methods by which this renewal period may be found via examples. However, prior to the solution of this example an explanation of the calibration of the model is offered, and issues that may arise are considered.

Equation 7.8 in the previous chapter identifies the service cost of assets: minimization of this function yields the optimal renewal period of the respective asset. This section highlights the parameters that need to be estimated, econometric estimation techniques, and possible violations of the standard Gaussian assumptions of the classical linear regression.

There are five parameters in Eq. 7.8 that *need to estimated:* $M(x)$, $S(x)$, $F(x)$, d, and I. These may take on a functional form or may be constant. Generally, d and I would be constants and $S(x)$, $F(x)$ and $M(x)$ would be functions. The discount rate, d, would be the discount rate that the utility uses for cost benefit analysis. A discount rate may be based on the rate at which the utility borrows, or bond yields of an appropriate term. In either case, the utility should use a long-term rate since water assets generally last 25–100 years. The investment cost, I, would be determined by the utility by estimating the labor and materials involved in a given renewal and aggregating these costs.

The maintenance expense, $M(x)$, should be known by the utility. The utility would have a maintenance schedule for assets and should be able to compute the total of labor and material costs at any given age, as these are costs that occur with certainty. The failure cost, $S(x)$, and frequency of failure, $F(x)$, are parameters that represent the expected failure cost at each age. These would have to be monitored

M.H. Dore, *Global Drinking Water Management and Conservation*,
Springer Water, DOI 10.1007/978-3-319-11032-5_8

over an extended period prior to estimation. Upon collection of the necessary data, explained above, the three functions would need to be estimated via regression analysis. It is important to note that time periods in which no maintenance is necessary should be recorded as having a zero maintenance cost, whereas periods without failures should be omitted entirely from the dataset. This is because of the certain nature of maintenance and uncertain nature of failure. It should also be noted that certain types of assets would have no maintenance, such as pipes. In this case, the maintenance parameter drops out of Eq. 7.8 entirely.

These functions, $M(x)$, $S(x)$, and $F(x)$, can be estimated via standard Ordinary Least Squares regression techniques. Upon completion of the regression, a critical investigation into the violations of the Gaussian assumptions must be conducted, and any violations must be addressed. Section 6 provides case studies of three municipalities in British Columbia. In the case studies, three violations of the Gaussian assumptions were detected: multicollinearity, heteroscedasticity and non-normality. These violations may arise with similar datasets in other geographical areas and are therefore given special attention here.

Multicollinearity is a relatively minor problem for two reasons. If multicollinearity exists, OLS is still "best linear unbiased estimator" (BLUE)—the coefficients are not biased by this violation. The other reason that multicollinearity is a minor problem in this application is that the multicollinearity often occurs between slope and intercept dummies that communicate the same information. There are three ways to address this violation. The user may want to return to the theory and determine if there is a sound theoretical reason for including only one of the variables.[1]

Heteroscedasticity is a violation that must be and is addressed more directly.[2] While this exploration centers around linear regression techniques, it is possible, and even likely, that the data follow a nonlinear pattern. We can check this by conducting nonparametric regressions. This technique determines a coefficient for each observation. Since the coefficient represents the slope of the regression line, different values for different observations indicate nonlinearity in the model. This is best observed graphically by plotting the cumulative value of the coefficient at each observation. The cumulative value is plotted since the value at any given observation would be the sum of slopes at that observation and all prior ones. In the event

[1] An alternative to this approach is to conduct regressions that exclude one variable and then the other to determine if these variables are significant in the absence of the other with which it has a high correlation. If they are not significant in the absence of multicollinearity, then there may be statistical grounds for excluding these variables. If there are no theoretical or statistical grounds for removing any of the variables, then these variables can still be included in the model, because, as identified above, OLS is unbiased by multicollinearity.

[2] In the case studies that follow, this violation is addressed in two ways. One can compensate for heteroscedasticity by utilizing White's covariance matrix in the regression. This technique adjusts the standard error on the coefficient to help determine which variables are in fact significant. Another approach to addressing this violation is to conduct a Robust Least Absolute Error regression. This method is preferred if there are other violations, such as non-normality. However, no goodness of fit statistics are provided with this technique. Therefore, the Robust LAE regression should only be performed if the third violation identified above, non-normality, is present.

that early data is missing, the mean observation in first time period would be considered the intercept. Upon plotting these points, one would be able to detect nonlinearity in the model. If nonlinearity is present, then an appropriate functional form will have to be chosen that fits the data. In the case studies that follow, two functional forms are used: logistic S curves and sine curves.

8.2 Towards Solving the DSS

Upon determining the parameters and functions involved in the model, an approximate solution can be derived. This solution will be useful in a practical application since the solution will only be "approximate" in the sense that at some point the decimals must be rounded. However, rounding the fourth decimal of a number identifying the year at which to replace an asset does not sacrifice the quality of the analysis, as scheduling and other factors will not permit a utility to be much more precise in terms of renewal.

There are two methods by which one would derive this approximate solution: graphically and numerically. A graphical solution involves plotting the cost function over time and visually identifying the minimum present value of service cost as a function of the renewal length of the asset. Adjusting the range of time to reflect only the area to either side of the solution will assist in deriving the more appropriate numerical solution. This process can be iterated until a solution of the desired precision is obtained. The numerical solution involves the use of computerized mathematical packages to solve numerically for a minimum. Since the cost function involves integrals, the computerized program may be required to derive a numerical solution where it is not reasonable to do so algebraically. A mathematical software package can be used to differentiate the cost function with respect to time. This first derivative would be set to zero to find the minimum. Upon discovery of the minimum, the second derivative would be checked to ensure that the cost function is convex (i.e. the second derivative is negative). The next section utilizes an example to demonstrate both of these techniques.

8.3 Application of Risk into the DSS

This section makes a number of reasonable assumptions as to the parameters of the model in order to demonstrate the application of the risk incorporated DSS. Consider parameters as follows:

$$M(x) = x^2 \tag{8.1}$$

$$S(x) = x^3 \tag{8.2}$$

$$d = 0.1 \tag{8.3}$$

$$f(x) = \frac{1 + \sin\left(\frac{x}{\pi} - \frac{\pi}{2}\right)}{2} \tag{8.4}$$

$$I = 100 \tag{8.5}$$

Maintenance and failure costs are assumed to increase at an increasing rate. Therefore, functions representing these costs must have positive first and second derivatives. The discount rate is 10 percent and the investment cost is 100. The probability of failure is monotonically increasing, first at an increasing rate, then at a decreasing rate. This is due to the fact that the probability of failure asymptotically approaches 100 percent as the asset ages. Therefore, the first derivative of this function must be positive and the second derivative must be positive over some initial range; after that the derivative changes to negative.

The generalized time-dependent service cost was derived in Chap. 7 (Eq. 7.8). Utilizing the parameters outlined above (Eqs. 7.1–7.5), Eq. 7.8 becomes:

$$C(t) = \frac{\int_{1}^{t-1} x^2 e^{-0.1x} \mathrm{d}x + \int_{1}^{t-1} x^3 e^{-0.1x} \frac{1+\sin\left(\frac{x}{\pi} - \frac{\pi}{2}\right)}{2} \mathrm{d}x + 100e^{-0.1t}}{1 - e^{-0.1(t+1)}} \tag{8.6}$$

The objective is to find the time t that minimizes Eq. 8.6. This will determine the renewal interval that minimizes the service cost of the asset over an infinite horizon, discounted to present values. This can be done numerically or graphically.

8.3.1 A Numerical Solution

A numerical solution can be derived with the help of a mathematical software package. This is standard continuous optimization. The first derivative, with respect to t, must be set to zero to find the minima and maxima. Then t is isolated to determine the values of t for which the function is minimized or maximized. The second derivative must be checked to determine which of these solutions are minima and which are maxima. If the second derivative is positive, then the corresponding value of t is a minimum. If it is negative, then the point is a maximum.

In this case, Eq. 8.6 can be expanded to eliminate the integrals yielding:

$$\begin{aligned} C(t) = {} & (31915.78910 - e0.1 - 0.1t(28955 - 2895t - 145t2 - 5t3 + \cos(0.3184713376t \\ & - 1.888471338)(171.0764517 + 60.43230478t - 3.408162357t2 - 1.429096774t3) \\ & + \sin(0.3184713376t - 1.888471338)(-136.0394311 + 40.37900330t + 12.39192863t2 \\ & - 0.4487363871t3)) + 100e - 0.1t)/(1 - e - 0.1t - 0.1) \end{aligned} \tag{8.7}$$

Equation 8.7 is differentiated to determine the first-order conditions for a minimum (the resulting equation is too large to display here). Since the maintenance cost clearly exceeds the investment cost after $t = 10$, the range over which the first derivative can be set to zero must be between 0 and 10. Setting the first derivative equal to zero and solving for t yields $t = 4.5099$ (after rounding). Checking the second order condition yields 39.2497—the function is convex at $t = 4.5099$, and therefore this value of t is the minimum. If the asset is renewed at $t = 4.5099$, then the service cost will be minimized.

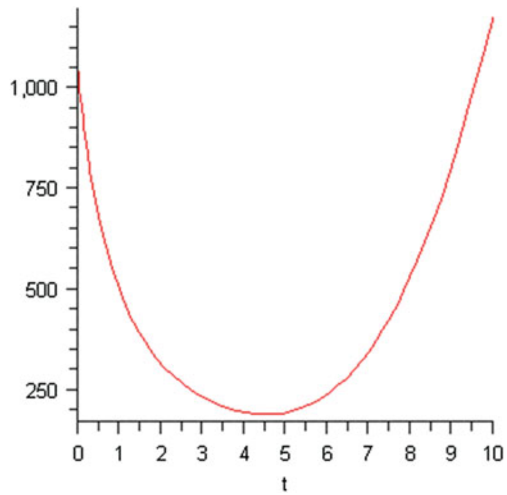
8.3.2 A Graphical Solution

An alternative approach to determining the optimal renewal period for assets is by plotting the cost function over an appropriate interval. In the example above, this interval would be $t = 0$–10, since maintenance exceeds the renewal cost after $t = 10$. The function is shown in Fig. 8.1:

It is obvious that the minimum occurs somewhere between $t = 4$ and $t = 5$. Through an iterative process the range of the graph is decreased until a value for t has been determined at an appropriate level of precision. Figure 8.2 demonstrates this iterative process and shows that the result is the same as with the numerical solution in Sect. 8.3.1.

This process demonstrates that the minimum cost of service occurs approximately at $t = 4.51$, i.e. the same as in Sect. 8.3.1. The asset in this example should be renewed at approximately every 4.5 years (assuming time is measured in years).

Fig. 8.1 The service cost of this asset is minimized if it is renewed approximately every 4–5 time periods



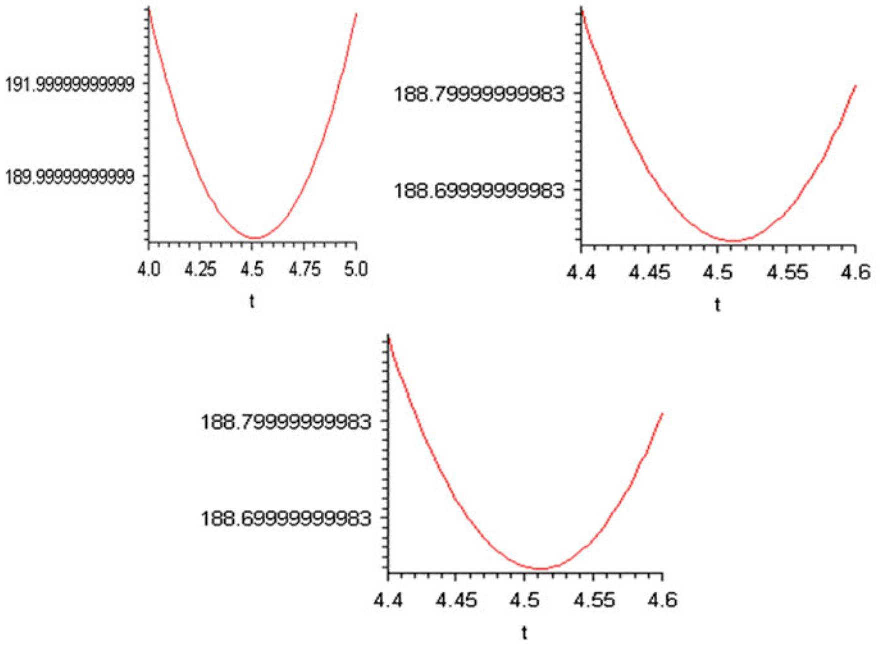


Fig. 8.2 Iterative adjustment reveals that the precise solution is 4.5 time periods

8.4 Case Studies from British Columbia

8.4.1 Introduction

This section utilizes data provided by municipalities in BC to estimate the parameters of the DSS model for purposes of determining the optimal renewal of water assets. The data supplied by the municipalities encompass a number of cities where the municipality is the wholesale supplier of water. Three of these cities are large enough that adequate sample sizes exist for the purposes of regression analysis. In order to preserve their privacy, these cities will be called cities A, B, and C. The data provide a record of failures over the period 1986–2005. This record indicates the date, cost, pipe size, pipe type, soil type, and depth of pipe. Since there are only two pipe types and two soil types, dummy variables will be used for these. Both slope (in conjunction with age) and intercept dummies were created. Additionally, the primary variable of concern is the age of the system, which was not included in the dataset. To determine the age, a survey of the housing market was conducted in each area. A more thorough explanation of the data manipulation can be found in the appendix along with all the regression results.

This application of the DSS determines the optimal renewal period for the whole network in each city. The parameters for the functions $S(x)$ and $F(x)$ are estimated via regression analysis. The discount rate was the 10-year bond yield and is given as 4.23 percent.[3] These networks are pipe networks, for which there is no maintenance expense. An approximate investment cost is determined for each city by taking half the value of all failure costs over the sample period. This is only an approximation for purpose of demonstration and compensates for data deficiencies. Lastly, when variables are found not to be statistically significant, they take on the mean value during solution of the model. This is because the model gives the optimal renewal of the network as a whole.

8.4.2 City A

The replacement cost of the network of City A will be set at half of the sum total of all the expenses arising from failure from 1986 to 2005. This amount is $41,278.

8.4.2.1 Determination of S(x)

In order to determine $S(x)$, the cost of failure was regressed on all the variables collected from the regional municipality. After correcting for multicollinearity, heteroscedasticity and non-normality, the significant variables were identified and the following Eq. 8.1 was estimated for the cost of failure to be used in the solution:

$$S(x) = -6362 + \underset{(7.05)}{673.38}\ \text{Depth}_i + \underset{(3.278)}{126.19}\ \text{Age}_i \tag{8.8}$$

8.4.2.2 Determination of F(x)

The function representing the frequency of failure for City A network was determined by regressing frequency of failure on age. It turned out that non-normality was again an issue. To resolve non-normality, a Robust LAE regression was performed. The following Eq. 8.2 for age was estimated:

$$F(x) = -7.5947 + \underset{(0.0788)}{0.25789}\ \text{Age}_i \tag{8.9}$$

8.4.2.3 Solution of the DSS

In the above regression analysis, the two most important parameters of the DSS were determined: Eqs. 8.8 and 8.9. These two modular parameters can now be added to the DSS in order to determine the optimal renewal period for the water network in City A. The cost-minimizing model from Sect. 7.2.3 is Eq. 7.8:

[3] This was the 10-year bond yield in 2007; but this can be easily varied.

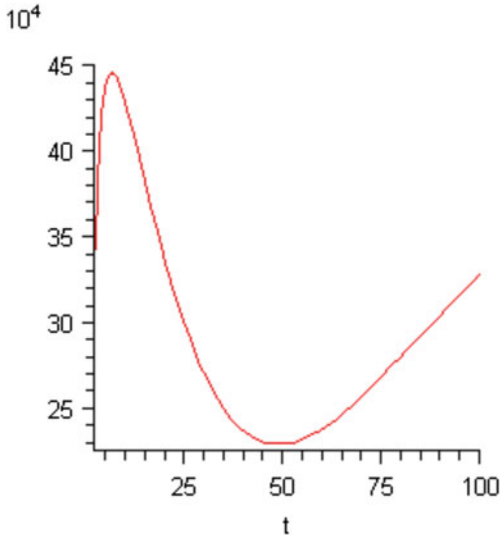


Fig. 8.3 The service cost of this asset is minimized if it is renewed approximately every 45–55 years

Substituting Eqs. 8.8 and 8.9, the investment cost and the discount rate, into Eq. 8.10 and assuming the average depth of 3.85 for all pipes in the network yields Eq. 8.3, which is depicted in Fig. 8.3:

$$C(t) = \frac{\int_{1}^{t-1} (-3769.487 + 126.19x)(-7.5947 + 0.25789x)e^{-0.0423x}\mathrm{d}x + 41278.46e^{-0.0423t}}{1 - e^{-0.0423(t+1)}} \tag{8.10}$$

Iterative scaling of this plot yields a minimum at approximately 48.7 years, which is consistent with the solution obtained via continuous optimization of 48.691, as shown in Fig. 8.4.

8.4.2.4 Nonlinearity

The nonparametric regressions of the functions $S(x)$ and $F(x)$ confirm that these functions are nonlinear as demonstrated in Figs. 8.5 and 8.6.

Figure 8.5 shows that $S(x)$ increases at a decreasing rate; Figure 8.6 shows that $F(x)$ is also increasing, at first at an increasing rate, then at a decreasing rate. Logistic curves will be used to approximate the curves in these diagrams. The rationale behind using logistic curves is that failure and cost of failure in the first few years is not expected to be very high. However, after a few years these amounts begin to grow. Yet, as the diagrams above show, there is a limit to this growth.

For the function $S(x)$, the curve seems to be approaching its carrying capacity, and so this value is set to 3,500. We do not expect very high costs of failure when

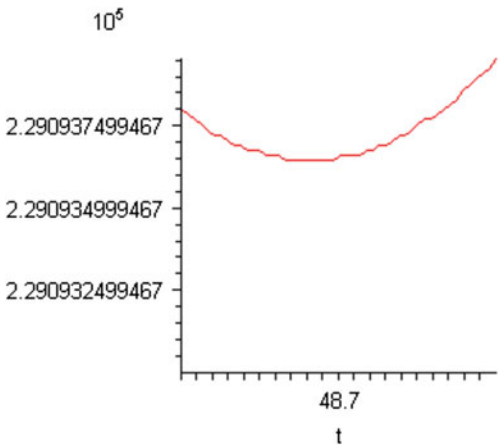


Fig. 8.4 Iterative adjustment reveals that the precise solution is 48.7 years

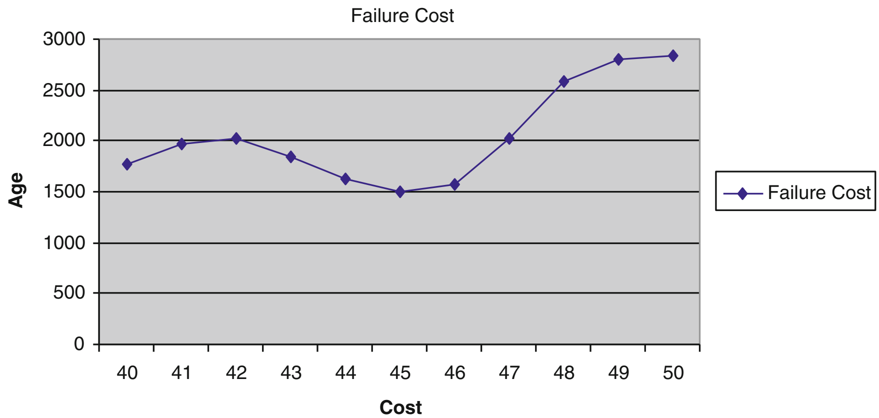


Fig. 8.5 $S(x)$ in City A is nonlinear

the asset is new, and so this value is set to 10. The growth rate is adjusted so that the logistic curve fits the diagram above as closely as possible and this value is found to be 0.15. This results in Eq. 8.11:

$$S(x) = \frac{35000e^{0.15x}}{3500 + 10(e^{0.15x} - 1)} \tag{8.11}$$

This function is depicted in Fig. 8.7.

The frequency of failure seems to have a carrying capacity of 4.5 and the initial value should be very small—it is set to 0.0001. Again, the growth rate is adjusted so

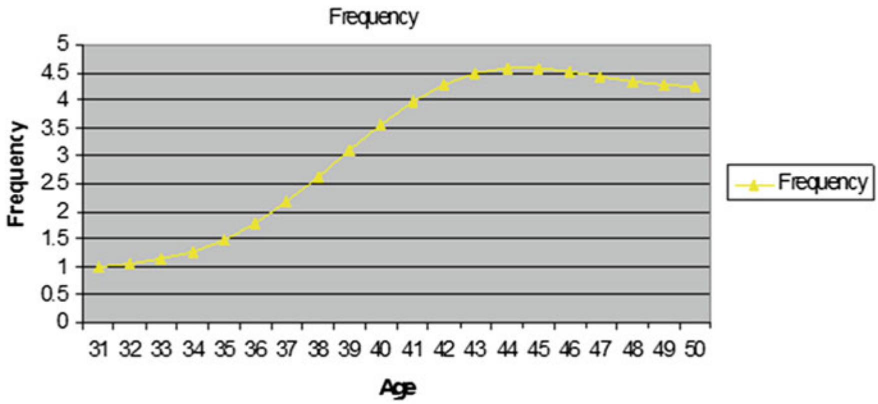
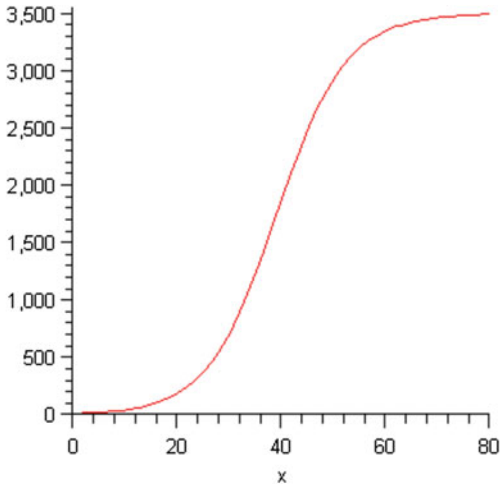


Fig. 8.6 $F(x)$ in City A is nonlinear

Fig. 8.7 A nonlinear, logistic S function approximating $S(x)$ in City A



that the logistic function fits the nonparametric regression and this yields a value of 0.31. This results in Eq. 8.12:

$$F(x) = \frac{0.00045e^{0.31x}}{4.5 + 0.0001(e^{0.31x} - 1)} \tag{8.12}$$

This equation is displayed in Fig. 8.8.

Utilizing these functions and in conjunction with the discount rate and investment cost from above, a new cost of service function is derived:

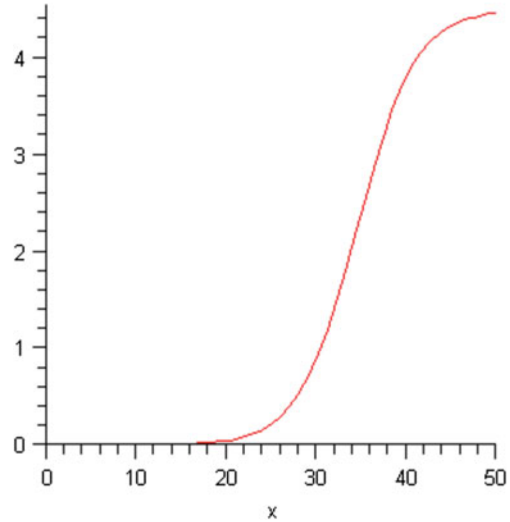


Fig. 8.8 A nonlinear, logistic S function approximating $F(x)$ in City A

$$C(t) = \frac{\int_{1}^{t-1} \left(\frac{35000e^{0.15x}}{3500+10(e^{0.15x}-1)}\right)\left(\frac{0.00045e^{0.31x}}{4.5+0.0001(e^{0.31x}-1)}\right)e^{-0.0423x}\mathrm{d}x + 41278.46e^{-0.0423t}}{1 - e^{-0.0423(t+1)}} \quad (8.13)$$

Continuous optimization yields a solution of 34.89, which is consistent with the graphical solution shown in Fig. 8.9.

8.4.3 City B

Techniques similar to those used for City A were also used for City B. It should be noted that the investment cost for City B is set at $119,178, which is half of the total of all failure costs as explained in the introduction to this chapter.

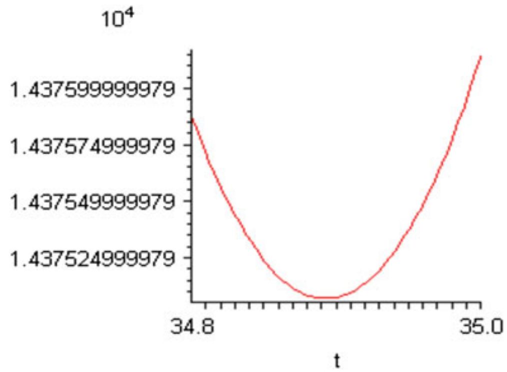


Fig. 8.9 The service cost of this asset is minimized if it is renewed approximately every 34.9 years

8.4.3.1 Determination of S(x)

Preliminary OLS regression analysis utilizing all available variables yielded no significant variables, very low adjusted R-squared, but a positive result for overall significance. An investigation into possible violations of the Gaussian assumptions reveals that multicollinearity, heteroscedasticity, and non-normality were all present.

A second regression was performed utilizing White's matrix to eliminate heteroscedasticity. This regression revealed that the only significant variables were age and depth. In order to correct for non-normality, a Robust Least Absolute Error (LAE) regression was performed. This final regression confirmed that age and depth are significant and resulted in Eq. 8.7 for cost of failure in City B:

$$S(x) = -1186.6 + \underset{(4.707)}{258.35}\ \text{Depth}_i + \underset{(2.885)}{49.322}\ \text{Age}_i \tag{8.14}$$

8.4.3.2 Determination of F(x)

The frequency of failure equation is determined by regressing frequency of failure on age. The results of this regression were significant. After correcting for heteroscedasticity and non-normality, a Robust LAE regression yielded Eq. 8.15 for frequency of failure in City B:

$$F(x) = -10.846 + \underset{(2.688)}{0.76923}\ \text{Age}_i \tag{8.15}$$

8.4.3.3 Solution of the DSS

The parameters of the DSS model for City B are as follows: Eqs. 8.14 and 8.15, $I = 119{,}178$ and $d = 0.0423$. The average depth of 3.24 is assumed for the network. Substituting these parameters into Eq. 7.8 results in Eq. 8.16:

$$C(t) = \frac{\int_{1}^{t-1} (-349.546 + 49.322x)(-10.846 + 0.76923x)e^{-0.0423x}\mathrm{d}x + 119178e^{-0.0423t}}{1 - e^{-0.0423(t+1)}} \tag{8.16}$$

Continuous optimization of this model yields a solution of 26.66. Iterated graphical minimization yields a consistent solution as depicted in Eq. 8.16 (Figs. 8.10 and 8.11).

8.4.3.4 Nonlinearity

Nonparametric regressions were performed to determine whether the cost and frequency functions in City B were nonlinear. The results have been shown in Figs. 8.12 and 8.13.

Fig. 8.10 The service cost of this asset is minimized if it is renewed at approximately every 20–35 years

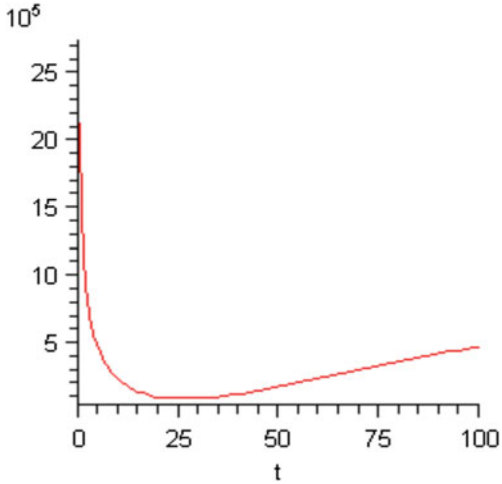
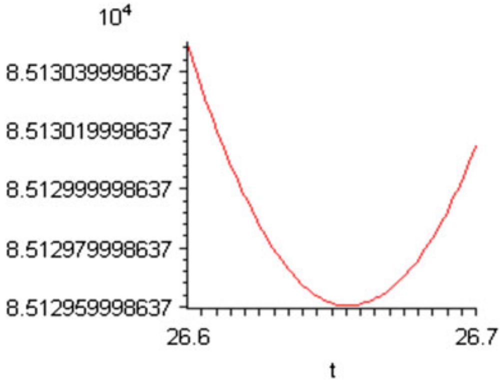


Fig. 8.11 Iterative adjustment reveals that the precise solution is 26.65 years



As in City A, logistic curves are fitted to the data shown in Figs. 8.12 and 8.3 to demonstrate initial slow growth that increases, and then slows again. The carrying capacity for the failure is set at 2,250 and the initial value is set at 10. Experimenting with different growth rates yields an ideal growth rate for this application of 0.21. Figure 8.14 demonstrates this.

The frequency of failure seems to have a carrying capacity of 25 and the initial value is again set low, this time at 0.01. Adjusting the growth rate to ensure that the curve fits the data yields a growth rate of 0.32. This function is depicted in Fig. 8.15. Utilizing these two new functions for cost and frequency yields Eq. 8.10.

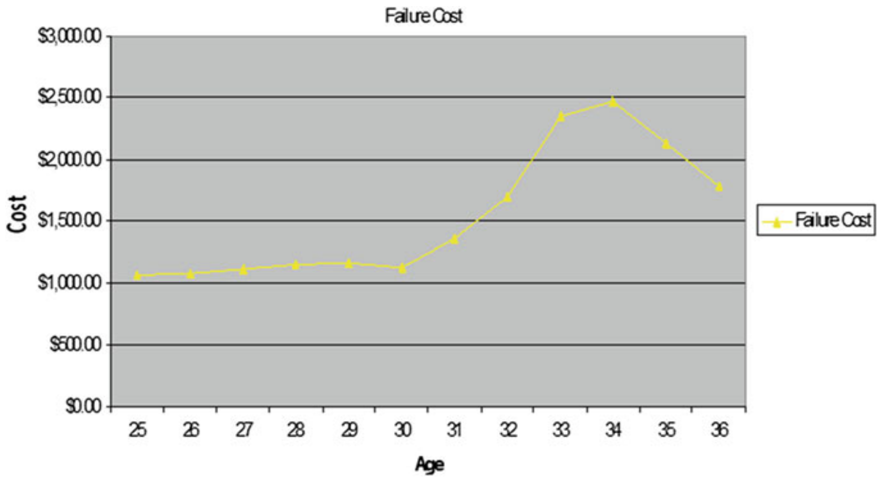


Fig. 8.12 $S(x)$ in City B is nonlinear

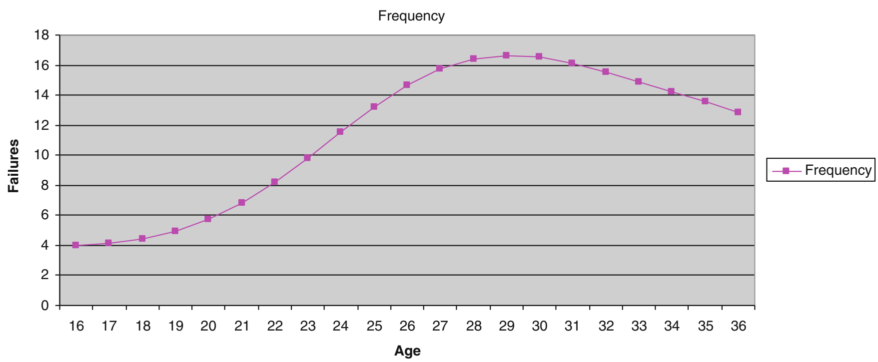


Fig. 8.13 $F(x)$ in City B is nonlinear

$$C(t) = \frac{\int_{1}^{t-1} \left(\frac{22500e^{0.22x}}{2250+10(e^{0.22x}-1)}\right)\left(\frac{0.25e^{0.32x}}{25+0.01(e^{0.32x}-1)}\right)e^{-0.0423x}\mathrm{d}x + 119178e^{-0.0423t}}{1-e^{-0.0423(t+1)}} \tag{8.17}$$

Continuous optimization of Eq. 8.17 yields a solution of 24.39. This solution is confirmed by iterative graphical minimization as demonstrated in Figs. 8.16 and 8.17.

8.4.4 City C

In City C, the same techniques as in Cities A and B were utilized. However, in this city, the results were very different both in the linear and nonlinear analysis. The

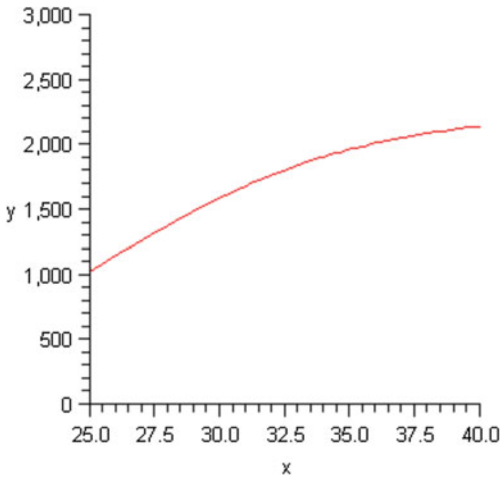


Fig. 8.14 A nonlinear, logistic *S* function approximating *S*(*x*) in City B, over the range 25–40

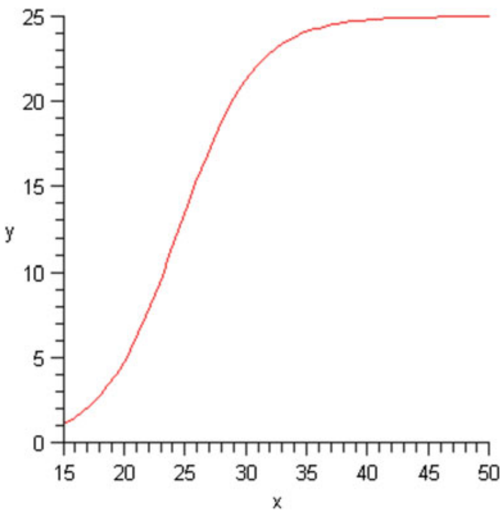


Fig. 8.15 A nonlinear, logistic *S* function approximating *S*(*x*) in City B, over the range 15–50

investment cost in this city is $152,169, which is determined by taking half the sum total of failure costs during the sample period.

8.4.4.1 Determination of S(x)

The cost of failure in City C was regressed on all available variables. However, only size and depth variables were significant. After correcting multicollinearity, heteroscedasticity, and non-normality, a Robust LAE regression was then performed utilizing all available variables and this revealed that all the variables, with the

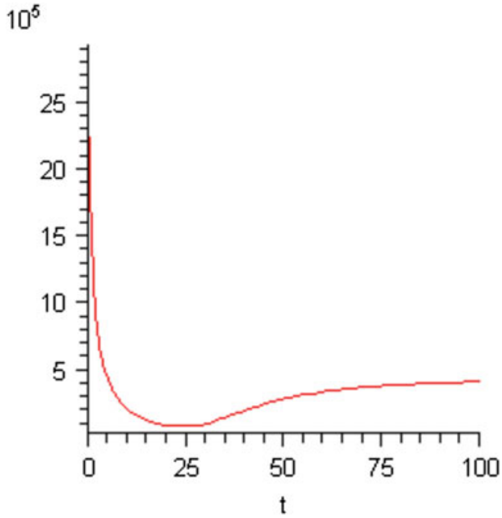


Fig. 8.16 The service cost of this asset is minimized if it is renewed approximately every 20–30 years

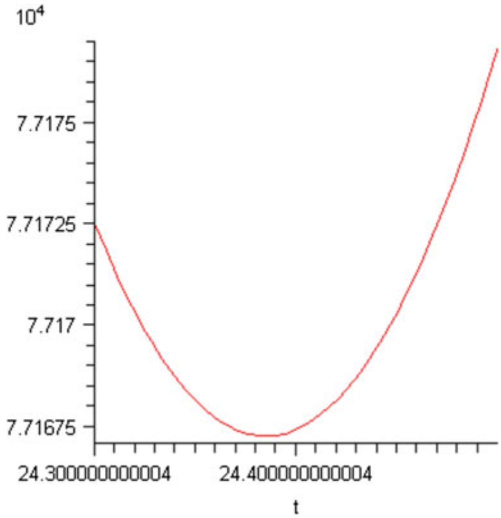


Fig. 8.17 Iterative adjustment reveals that the precise solution is 24.4 years

exception of the soil type and copper pipe variables were significant. Upon removal of the insignificant variables, Eq. 8.18 for cost of failure was estimated:

$$\begin{aligned} S(x) = {} & -3661.8 + \underset{(4.414)}{230.49}\ \text{Depth}_i + \underset{(4053)}{165.7}\ \text{Size}_i - \underset{(-4.556)}{296.01}\ \text{ACAGE}_i \\ & + \underset{(4.493)}{13319}\ \text{AC}_i + \underset{(3.124)}{87.05}\ \text{Age}_i \end{aligned} \tag{8.18}$$

8.4.4.2 Determination of F(x)

To determine the frequency of failure in City C, frequency was regressed on age. This initial regression (6.4.2a) provided significant results. Again after correcting for heteroscedasticity, a second regression, utilizing White's matrix also demonstrated significant results. Since non-normality was not an issue in this second regression, these results were used to obtain Eq. 8.19, which is the frequency function for City C:

$$F(x) = -13.587 + \underset{(2.341)}{0.4987}\ \text{Age}_i \tag{8.19}$$

8.4.4.3 Solution of the DSS

The DSS parameters in City C are: Eqs. 8.18 and 8.19, d = 0.0423 and I = 152169.31. Average values were assumed for depth, size, ACAGE and AC, resulting in Eq. 8.13, a modified cost of failure function:

$$S(x) = 209.83 + 87.05\ \text{Age}_i \tag{8.20}$$

These parameters yield Eq. 8.21, a DSS for City C:

$$C(t) = \frac{\int_1^{t-1} (209.83 + 87.05x)(-13.587 + 0.4987x)e^{-0.0423x}\mathrm{d}x + 152169.31e^{-0.0423t}}{1 - e^{-0.0423(t+1)}} \tag{8.21}$$

Iterated graphical minimization yields an interesting solution in this case as demonstrated in Figs. 8.18 and 8.19.

Figure 8.19 seems to indicate that the net service cost can be negative if the asset is renewed every 30.2 years. Continuous optimization confirms this result. Of course, a negative cost does not make sense. This result demonstrates the limitations of linear analysis. A simple examination of the parameters demonstrates that the number of failures in the early years of the asset is actually negative, which results in a negative expected failure cost and this obviously does not make any sense.

8.4.4.4 Nonlinearity

Nonparametric regressions for City C demonstrate that the frequency of failure follows a similar pattern to Cities A and B. However, the cost of failure does not seem to behave as in the other two municipalities. Rather than resemble a logistic S function, the cost of failure in City C seems to resemble a sine function. The results of the nonparametric regressions are shown in Figs. 8.20 and 8.21.

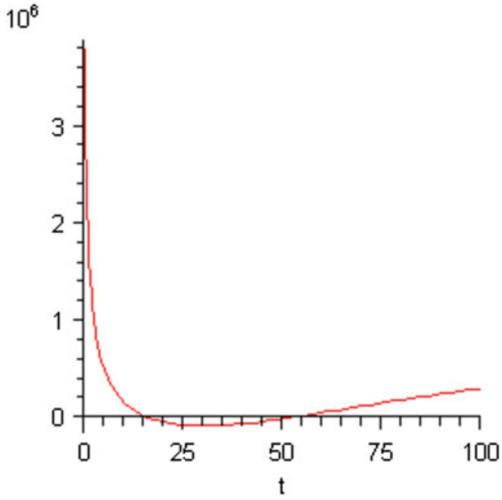


Fig. 8.18 The service cost of this asset is minimized if it is renewed approximately every 25–35 years. However, this value appears to be negative

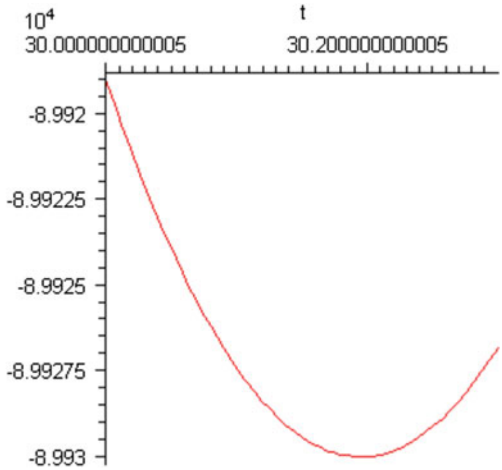


Fig. 8.19 Iterative adjustment reveals that the precise solution is 30.2 years

Since the cost function resembles a sine wave, the function $y = \sin(x)$ was adjusted to fit the nonparametric regression. First, a value of 5,100 was added to the right side of the equation so that the function would oscillate above and below the mean. Next, we multiply the sine term by 300 to ensure that these oscillations occur between 4,800 and 5,400. Lastly, the term inside the brackets was changed to $\frac{\pi}{4}x - 1.5$ to ensure that the cycle time was 8 years and to ensure that the minima and maxima coincided with the nonparametric regression. These manipulations resulted in Eq. 8.22 for cost:

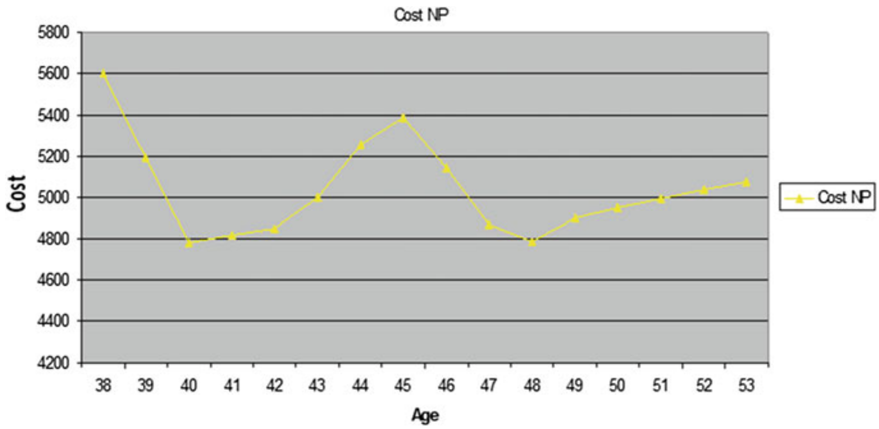


Fig. 8.20 $S(x)$ in City C is nonlinear

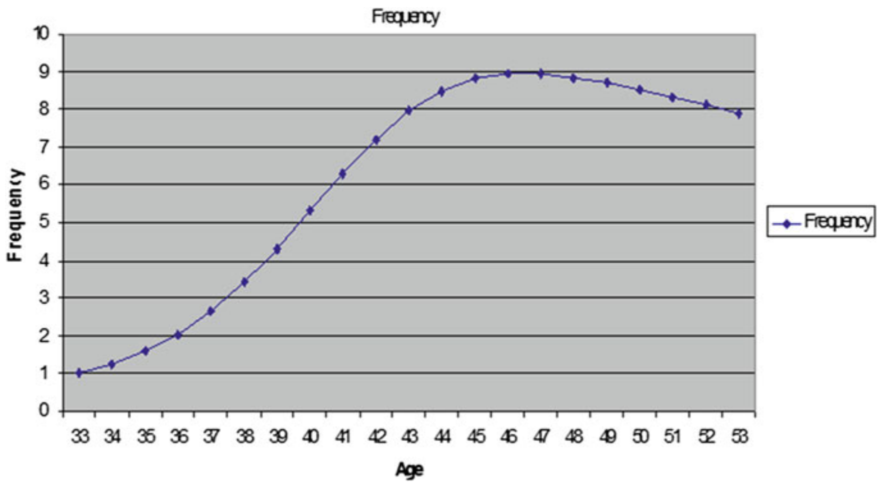


Fig. 8.21 $F(x)$ in City C is nonlinear

$$S(x) = 5100 + 300\sin\left(\frac{\pi}{4}x - 1.5\right) \tag{8.22}$$

As in the other municipalities, the frequency function resembles a logistic curve. In City C, this logistic curve seems to have a carrying capacity of 10. The initial value was set at 0.0001 and the growth rate was adjusted until it best fit the model. This yielded a growth rate of 0.29 and Eq. 8.23 for frequency:

$$F(x) = 0.001 \frac{e^{0.29x}}{10 + 0.0001(e^{0.29x} - 1)} \tag{8.23}$$

Utilizing Eqs. 8.22 and 8.23 in conjunction with the other parameters for City C yields Eq. 8.24, the DSS cost function for this municipality is:

$$C(t) = \frac{\int_{1}^{t-1} \left(5100 + 300\sin\left(\frac{\pi}{4}x - 1.5\right)\right)\left(\frac{0.001e^{0.29x}}{10+0.0001(e^{0.29x}-1)}\right)e^{-0.0423x}\mathrm{d}x + 119178e^{-0.0423t}}{1 - e^{-0.0423(t+1)}} \tag{8.24}$$

Continuous optimization of Eq. 8.24 yields a minimum at 35.03 and iterative graphical minimization yields a consistent result as demonstrated in Figs. 8.22 and 8.23:

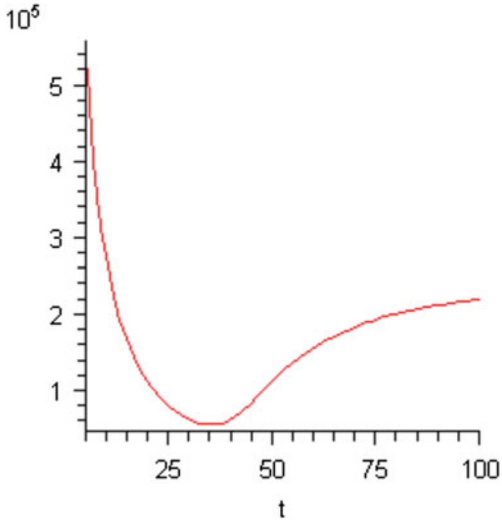


Fig. 8.22 The service cost of this asset is minimized if it renewed at approximately every 30–40 years

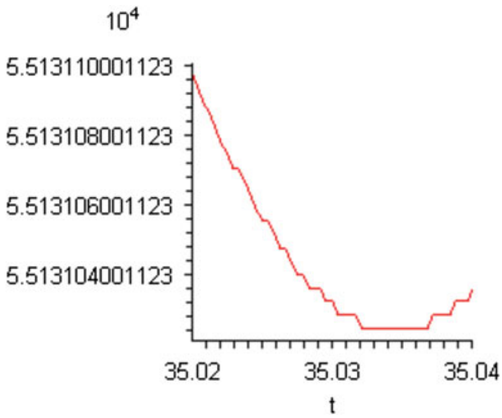


Fig. 8.23 Iterative adjustment reveals that the precise solution is 35.03 years

8.5 Conclusion

This chapter demonstrates the usefulness of the DSS model. The cost and frequency functions for three different cities were estimated by two methods and then used to determine the optimal renewal period via the DSS for each method. In City A, the cost function behaved in an intuitive manner. However, the linear estimation techniques resulted in a considerably different solution from the nonlinear techniques. Linear estimation results in an optimal renewal period of 48.7 years, whereas the nonlinear estimation results in a renewal of 35.9 years. In City B, the cost functions again behaved in an intuitive manner. This time, however, the linear and nonlinear estimation techniques produced very similar results in terms of the optimal renewal time. Linear analysis resulted in an optimal renewal of 26.7 years whereas nonlinear analysis resulted in an optimal renewal of 24.4 years.

The linear estimation techniques resulted in illogical results when used in City C. The DSS in this city was minimized at a renewal period of 30.2 years. However, the present value of the cost of service was negative at this renewal period—an impossible result. The nonlinear techniques, however, did not exhibit this illogical behavior in City C. Interestingly, the optimal renewal period under nonlinear techniques is found to be 35 years.

There are, however, limitations that need to be mentioned. The data collected from the Regional Municipality involved only a 20-year period. Therefore, the behavior of the relevant functions before and after this period cannot be estimated via regression analysis. This behavior can only be inferred based on theory or the behavior during the period for which data exist. Another limitation of the analysis stemming from the data involved the incomplete nature of data. At times, only a portion of the relevant information was available. This seemed to occur more frequently with the earlier data. Also, there were years where no failures were recorded and these years sometimes fell in between years of numerous failures. It is possible that due to the large costs of data collection, or due to personnel changes, some data were not collected for these years. The last data limitation that should be mentioned pertains to what the data includes. It was not possible to determine if the cost included peripheral damages, and on occasions the cause of failure was not known. This model assumes that failure occurs as a result of "normal" wear-and-tear use under "normal" conditions. However, if a pipe had been installed incorrectly or if a 100-year storm had occurred, then the optimal solution may not be correct. The age at which failure occurs was also not given in the dataset and a benchmark had to be established for each city to approximate a date of construction for each distribution network as a whole. This is only an estimate and assumes that the whole network was built at the same time.

As with all applied economic research, the data limitations identified above should be taken into account. Nevertheless, the DSS demonstrates its power to incorporate risk in managing water infrastructure assets.

In this application, the DSS was used to determine the optimal renewal for the whole network. This is not the only way in which the DSS may be used. If enough detailed information existed, the DSS could be used for any sort of configuration. It could be used for small, medium, or large-scale renewals. The case studies examined network renewal because there was not enough information to work on a smaller scale. Logically, the different parts in the network will need to be replaced at different intervals. Therefore, the application of the DSS will only improve in the future as the dataset that utilities collect both improves and grows.

The main practical lesson that emerges from the analysis of asset management is that it is cost-effective to take risk into account and that taking risk into account means *renewing or replacing* the equipment *before* it breaks down or causes a serious failure of the water supply infrastructure network.

Chapter 9
Threats to Human Health: Use of Chlorine, an Obsolete Treatment Technology

9.1 Introduction

Although chlorine is not the only disinfecting agent available to the water supply industry, it is the most widely used disinfectant in North America. It is currently employed by over 98 percent of all US water utilities that disinfect drinking water (Calomiris and Christman 1998). However, it is ineffective against parasitic protozoans *Cryptosporidium parvum* and *Giardia lamblia*. The use of chlorine as a disinfectant has one major drawback. Disinfection byproducts (DBPs) are formed through chemical reaction between natural organic matters (NOMs) and the disinfectant (i.e. chlorine, chloramine and chlorine dioxide) in the treatment of drinking water. Chlorinated DBPs have been recognized as a potential public health concern in drinking water since DBPs were first reported in the1970s[1] and identified as a carcinogen in 1976 (National Cancer Institute 1976). Since then, more than 700 chemical compounds associated with DBPs such as trihalomethanes (THMs) and haloacetic acids (HAAs) have been identified, which are estimated to account for approximately 50 percent of the total organic halides (TOX) formed by chlorination (Villanueva et al. 2014). As the main DBPs, THMs make up around 20–30 percent of TOX, and they are the most commonly regulated (Itoh et al. 2011). In particular, the maximum acceptable concentration of total THMs in the European Union (EU), Canada, and Ontario is 100 μg/L, but in Ontario a further reduction to 80 μg/L was under active consideration (in 2008–2009), to bring it in line with the USEPA, which also has a maximum Acceptable Concentration (MAC) of 80 μg/L. However, as of May 2014, the MAC in Ontario and Canada is still 100 μg/L.

It is clear that the long-term health risks associated with the use of chlorine are being recognized. However, the sampling requirement for THMs is somewhat lax: samples are supposed to be a “running annual average” or a moving average of the four past quarters; a single result that exceeds the 100 μg/L is not interpreted as

[1] In 1974, Rook (1974) first discovered DBPs in the Netherlands.

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exceeding the maximum acceptable concentration. To date, there are a total of 18 DBPs for which the World Health Organization (WHO 2006), USA (USEPA 2006), European Union (1998), Canada (Health Canada 2012) and Japan (Japanese Council on Public Welfare Science 2003) have set health-based guideline values (Itoh et al. 2011). Yet, there is no guideline of the tolerable level of DBP that would avoid developmental and reproductive toxicity (Itoh et al. 2011). DBPs are complex mixtures, but current management practices focus on meeting the maximum concentration levels (MCLs) for individual DBPs. Therefore, current water quality management is insufficient to reduce overall toxicity of DBPs (Villanueva et al. 2014).

In this chapter, we focus on exploring the long-term health effects of using chlorine in untreated water. Section 9.2 reviews a select set of epidemiological studies on some main areas of health impacts in humans from exposure to DBPs, including cancer (mainly bladder cancer), adverse reproductive and developmental outcomes, blood lead levels, as well as estrogenic effects. Section 9.3 discusses the current management practices in some developed countries, including USA, Canada, and some EU countries, such as Germany, Denmark, and the Netherlands.

9.2 Long-Term Health Effects of Using Chlorine

9.2.1 Chlorinated DBPs Exposure with Cancer Incidence

Chlorinated drinking water contains a complex mixture of chlorinated and brominated byproducts with mutagenic and carcinogenic properties; toxicologists have known this for a long time. A number of studies have drawn an association between the consumption of chlorinated drinking water and cancer due to the DBPs (i.e. THMs and HAAs). The adverse effects of DBPs are not universally supported, partly because the effects can vary in time and space (Villanueva et al. 2012). As a matter of fact, the epidemiological studies of DBP exposures and health effects in humans have focused on a small subset of the several hundred DBPs that may occur in public water supplies (Richardson et al. 2007); they have focused primarily on THMs and HAAs (Hinckley et al. 2005, Hoffman et al. 2008, and Righi et al. 2012).

An early WHO (2004) report was inconclusive although it did carry some evidence of adverse health effects, particularly due to chloroform, one of the common THMs. But more recent findings suggest there is cause for concern. As summarized in Table 9.1, a series of research on potential carcinogens reviewed by the WHO International Agency for Research on Cancer (IARC) indicated that there is evidence on the carcinogenicity of DBP compounds in drinking water. First, as primary THMs, Chloroform (IARC 1999) and bromodichloromethane (BDCM) (IARC 1991) are classified as possible human carcinogens, and have been linked to reproductive defects in animal studies, while Dibromochloromethane (DBCM) and bromoform are not yet so classified, indicating there is no evidence supporting these two compounds as carcinogens, but there is insufficient evidence to classify them as

Table 9.1 Evidence of carcinogenicity as concluded by the IARC for some chemicals whose main pathway of human exposure is through drinking water (Villanueva et al. 2014)

Compounds	Human evidence	Animal evidence	Overall evaluation[a]	IARC Monograph
DBPs*: *Trihalomethanes				
Chloroform	Inadequate	Sufficient	2B	Vol. 73 (IARC 1999)
Bromodichloromethane	Inadequate	Sufficient	2B	Vol. 52 (IARC 1991)
Dibromochloromethane	Inadequate	Limited	3	Vol. 52 (IARC 1991)
Bromoform	Inadequate	Limited	3	Vol. 52 (IARC 1991)
DBPs*: *Haloacetic acids				
Dichloroacetic acid	Inadequate	Sufficient	2B	Vol. 106 (IARC 2013)
Trichloroacetic acid	Inadequate	Sufficient	2B	Vol. 106 (IARC 2013)
Bromochloroacetic acid	Inadequate	Sufficient	2B	Vol. 101 (IARC 2012)
Dibromoacetic acid	Inadequate	Sufficient	2B	Vol. 101 (IARC 2012)

[a] Group 1 (the agent is carcinogenic to humans), 2A (the agent is probably carcinogenic to humans), 2B (the agent is possibly carcinogenic to humans), 3 (the agent is not classifiable as to its carcinogenicity to humans)

noncarcinogenic (IARC 1991). Moreover, all four primary HAA compounds are classified as possible human carcinogens, and there is sufficient evidence from experimental animals for the carcinogenicity (IARC 2013). It should be noted that the classification of possible human carcinogens is extrapolated based on the data that has come from research on animals (IARC 2010). Hence, there is inadequate epidemiological evidence of carcinogenicity in humans for *all* DBPs compounds.

Furthermore, the USEPA (2003) has calculated the cancer potency factors for the four THMs in mg of chemical per kg of body weight per day (expressed as mg/kg/day). BDCM has the highest factor of 0.062 mg/kg/day, followed by Bromoform with 0.0079 mg/kg/day, and Chloroform with 0.0061 mg/kg/day, while DBCM is considered noncancerous. The cancer potency factors have some foundation in the study reported below.

9.2.1.1 Bladder Cancer

Cantor et al. (1987) summarized a number of case-control studies on the incidence of cancer that have been undertaken in North America, such as Western Maryland,

Table 9.2 Individual-based studies related to chlorination byproduct exposure (Cantor 1997)

Author (Reference) Year	Cancer sites (years of diagnosis)	Number of cases	Study location	Exposure timing
Wilkins and Comstock (1981).	Bladder, liver, kidney 1963–1975	81,45,31	Western Maryland (USA)	Years in 1963 domicile
Cantor et al. (1987)	Bladder 1978	2,805	10 locations in USA	Lifetime
Freedman et al. (1997)	Bladder 1975–1992	294	Western Maryland (USA)	Years in 1975 domicile
McGeehin et al. (1993)	Bladder 1988–1989	327	Colorado (USA)	Age 20 to interview
Cantor (1997)	Bladder 1986–1989	1,452	Iowa (USA)	Lifetime
Marrett et al. (1996)	Bladder 1992–1994	927	Ontario (Canada)	

Note In all studies, individual histories of water source and chlorine disinfection were developed by combining residential information from the questionnaire with historical data from water utilities

Iowa, Colorado, and Ontario (see Table 9.2). He pointed out that "the evidence for carcinogenicity of chlorination byproducts is strongest for bladder cancer, where associations were found overall or in major subgroups in five case-control studies and one population cohort study" (Cantor 1997).

Villanueva et al. (2003) carried out a meta-analysis of the best available epidemiological evidence on chlorinated drinking water and bladder cancer. This meta-analysis included six case studies (including one from Ontario) and two cohort (panel data) studies. We begin by restating the results of one of the panel data studies, that of Wilkins and Comstock 1981. They found that for both sexes the probability of getting bladder cancer from drinking chlorinated water was 70 percent higher than from drinking deep well waters—that is what an odds ratio of 1.7 means. The breakdown by gender is shown in Table 9.3.

The results of the meta-analysis (summarized in Table 9.4) show that the odds ratio increases with exposure to chlorinated drinking water, from 1.13 (or the probability is 13 percent higher due to chlorinated water) after 20 years to 1.43 after 60 years, i.e. the probability of bladder cancer is 43 percent higher after 60 years of exposure to chlorinated drinking water.

Table 9.3 Odds Ratios from Wilkins and Comstock (1981)

	Deep well users	Chlorinated surface water users
Men	1.0	1.8 (CI: 0.8–4.75)
Women	1.0	1.60 (CI: 0.54–6.32)
Both sexes	1.0	1.7 (CI: 0.8–3.5)

Odds Ratios (OR) and 95 percent Confidence Intervals (95 percent CI)

Table 9.4 Odds Ratios estimated in Meta-Analysis (Villanueva et al. 2003)

Combined unit increase	Slope	Standard error	Odds ratio	95 percent C. I.
	0.006	0.000128	1.006	1.004–1.009
20 years			1.13	1.08–1.20
40 years			1.27	1.17–1.43
60 years			1.43	1.27–1.72

Dose-response regression slopes obtained from weighted least squares within study, and OR with 95 percent CI obtained from the meta-analysis of the five slopes and their standard errors. Both sexes

Table 9.5 Pooled analysis of bladder cancer and THMs, by gender (Villanueva et al. 2004)

Odds ratios (ORs) with 95 percent Confidence Intervals		
Cumulative exposure to THMs (mg)	Male	Female
>0	1.30 (1.14–1.50)	1.06 (0.77–1.45)
0–15	1.00	1.00
>15–50	1.22 (1.01–1.48)	0.92 (0.65–1.32)
>50–400	1.28 (1.08–1.51)	0.94 (0.70–1.27)
>400–1000	1.31 (1.09–1.58)	1.02 (0.74–1.41)
>1000	1.50 (1.22–1.85)	0.92 (0.65–1.30)

Another very important analysis was done by Villanueva et al. (2004), who pooled the primary data from six case-control studies of bladder cancer in USA, Canada, France, Italy, and Finland, respectively, by using THMs as indicator of DBPs. They found that there was an odds ratio of 1.3 in men who were ever exposed to THMs in drinking water compared with those who were never exposed to THMs during the 40-year exposure window, while the odds ratio for women who were ever exposed to THMs was 1.06. The results indicate that there was an exposure-response relationship between DBPs intake and bladder cancer for men, but no relationship was observed in women. Moreover, the risk of bladder cancer for men can be increased by up to 50 percent when men are exposed to more than 1,000 mg THMs during the 40-year exposure window (Table 9.5).

Villanueva et al. (2004) further found that men exposed to chlorinated drinking water for 35–45 years had an increased risk of bladder cancer compared with those exposed for less than 5 years. That is, the probability of bladder cancer is 24 percent higher after at least 35 years of exposure to THMs, while the probability is 15 percent higher due to chlorinated water after 5–14 years. Similar results were found when the odds ratios are adjusted for gender, age, center, smoking status, education, ever worked in high-risk occupations, heavy coffee consumption (5 cups/day) and total fluid intake. Finally, Villanueva et al. (2004) stated that "these findings strengthen the hypothesis that the risk of bladder cancer is increased with long-term exposure to disinfection byproducts at levels currently observed in many industrialized countries" (Table 9.6).

Table 9.6 Association of Average Exposure to THMs Higher than 1 μg/L with Bladder Cancer, Within Specific Time Windows of Exposures for Men (Villanueva et al. 2004)

Time window before the interview	OR (95 percent CI)	OR (95 percent CI)[a] Adjusting for All Other Time Periods
5–14 years	1.15 (1.00–1.32)	1.05 (0.84–1.31)
15–24 years	1.19 (1.04–1.36)	0.92 (0.70–1.21)
25–34 years	1.29 (1.12–1.48)	1.22 (0.95–1.58)
35–45 years	1.24 (1.07–1.44)	1.13 (0.93–1.37)

[a] Odds Ratios are adjusted for sex, age, center, smoking status, education, ever worked in high-risk occupations, heavy coffee consumption (5 cups/day) and total fluid intake. The analysis was conducted only among subjects with 70 percent exposure information

9.2.1.2 Colon and Rectal Cancer

A study conducted in Iowa, USA from 1986 to 1987 by Doyle et al. (1997). assessed the relationship between chlorination DBPs in drinking water and colon cancer in women. They found that the exposure to chlorination DBPs in drinking water was associated with increased risk of colon cancer for women. These findings are consistent with some, but not all previous epidemiological and animal studies, and suggest that long-term exposure to chlorination DBPs in drinking water may be associated with an increased risk of cancer in humans (Doyle et al. 1997).

King et al. (2000) undertook a population-based case-control study in southern Ontario, Canada from 1992 to 1994 to assess the relationship between chlorinated byproducts in public water supplies and cancers of the colon and rectum by using the THMs as an indicator of DBPs. The analyses included 767 colon cases, 661 rectal cases, and 1,545 controls. For men, colon cancer risk was increased and associated with cumulative exposure to THMs in drinking water, and men exposed to chlorinated drinking water for 35 years had an increased risk of colon cancer compared with those who were exposed to THMs for less than 10 years. The probability of colon cancer reached 53 percent higher after at least 35 years of exposure to THMs (see Table 9.7). The cumulative THM exposure was associated with a 17 percent increase in risk for each 1,000 μg/L per year. Moreover, the long-term (at least 35 years) exposure to a THM level of 75 μg/L was associated with a doubled colon cancer risk in men, while these relationships were not observed among women. Furthermore, in the analysis of rectal cancer, no relationship was observed between rectal cancer risk and any of the measures of exposure to chlorination DBPs for either gender (see Table 9.7).

In contrast, Bove et al. (2007) conducted a case control study of 128 cases and 253 controls in Monroe County, Western New York State, USA from 1998 to 2003 to assess the effects of exposure to four primary THMs in drinking water on rectal cancer among white males by using a logistic regression, and found that increasing levels of three of the four primary THMs did correspond with an increase in risk for rectal cancer, although risk of rectal cancer did not increase with total level of

Table 9.7 Odds Ratios and 95 percent Confidence Intervals (CI) for risk of cancers of the colon and rectum according to exposure to water factors, by sex (King et al. 2000)

	Colon cancer		Rectal cancer	
Water factor level	Men OR (CI)	Women OR (CI)	Men OR (CI)	Women OR (CI)
Chlorinated (year)				
0–9	1.00	1.00	1.00	1.00
10–19	1.70 (1.07–2.68)	0.55 (0.32–0.94)	1.10 (0.69–1.76)	0.71 (0.40–1.27)
20–34	1.33 (0.96–1.86)	0.85 (0.58–1.24)	0.98 (0.71–1.36)	0.89 (0.58–1.37)
>34	1.53 (1.13–2.09)	0.74 (0.52–1.05)	0.97 (0.72–1.32)	1.04 (0.71–1.53)
THM > 75 μg/L (year)				
0–9	1.00	1.00	1.00	1.00
10–19	1.12 (0.85–1.46)	0.91 (0.66–1.25)	0.89 (0.67–1.18)	1.00 (0.71–1.40)
20–34	1.49 (0.99–2.26)	0.92 (0.54–1.56)	1.08 (0.70–1.68)	0.81 (0.44–1.47)
>34	2.10 (1.21–3.66)	1.20 (0.60–2.42)	0.96 (0.49–1.89)	0.89 (0.39–2.02)

THMs. The odds ratios for bromoform, DBCM, DCBM, and Chloroform were 1.85, 1.78, 1.15, and 1, respectively, indicating that the first three THM compounds lead to an increased risk of rectal cancer. In particular, exposure to bromoform in drinking water may be associated with the highest risk for rectal cancer.

9.2.2 Effects on Preterm Births and Health Defects in the Unborn Child

Exposures and risks affect vulnerable populations such as pregnant women and the unborn child, to which we now turn. In an important study, Lewis et al. (2007) focused on a single water utility during 1999–2001. Vital record data were obtained for a large, racially diverse population residing in 27 Massachusetts communities that received drinking water from a single public utility. This water system was monitored weekly for total THMs, and it maintained geographically stable total THMs levels system-wide during the study period. They employed proportional hazards regression to examine the effects of trimester-specific and shorter-term peak exposures to total THMs in drinking water late in pregnancy on preterm births in 37,498 singletons. They found that for all women, there was an increase in risk for delivering a preterm baby when exposed to ≥60 μg/L total THMs during the

4 weeks before birth [hazard ratio (HR) = 1.13; 95 percent confidence interval (CI), 0.95–1.35]. However, women who depended on a governmental source of payment for prenatal care were at increased risk when exposed at such levels late in gestation (HR = 1.39; 95 percent CI, 1.06–1.81). In contrast, exposure to high levels of total THMs during the second trimester and high exposure throughout pregnancy resulted in a 15–18 percent reduction in risk for preterm delivery in this population.

In addition to the risk of preterm births, a published Birmingham University study is even more disturbing: Hwang et al. (2008) suggest that drinking tap water containing total THMs while pregnant may double the risk of serious health defects in the unborn child. The authors conducted a population-based cross-sectional study of 396,049 Taiwanese births in 2001–2003 using information from the Birth Registry and Waterworks Registry. They compared the risk of eleven most common specific defects in four disinfection byproduct exposure categories based on the levels of total THM representing (1) high (total THMs 20 + μg/L), (2) medium (total THMs 10–19 μg/L), (3) low exposure (total THMs 5–9 μg/L), (4) very low exposure (0–4 μg/L) as the reference category. In addition, they also carried out a meta-analysis of the results from the present and previous studies focusing on the same birth defects. In summary, their results show:

(1) For the risk of ventricular septal defects, the adjusted odds ratio was 1.81, with the 95 percent confidence interval being 0.98–3.35 compared to the reference category;
(2) For cleft palate, the adjusted odds ratio was 1.56, with the 95 percent confidence intervals being 1.00–2.41, compared to the reference category;
(3) Anencephaly is a birth defect in which a baby is born without a major portion of the brain, skull, and scalp. The odds ratio for anencephaly was 1.96, with the 95 percent confidence being 0.94–4.07, compared to the reference category.
(4) In the meta-analysis, their summary odds ratio for ventricular septal defects was 1.59, with the 95 percent confidence intervals being 1.21–2.07.

9.2.3 Changes in Blood Levels

It is interesting that research indicates that exposure to showering and washing may be worse than drinking chlorinated water, if whole blood levels of THM are measured (Backer et al. 2000; Nuckols et al. 2005). Backer et al. found that the highest levels of THMs in the blood were found in people who took a ten minute shower and the lowest levels were found in blood samples from people who drank 1 liter of water over a 10 min period. Drinking of one liter of water increased blood THMs by less than 10 percent of the increase resulting from showering or bathing for 10 min. Like other volatile organic compounds, there is evidence that the THMs may bioaccumulate in the body (Ashley and Prah 1997).

Nuckols et al. confirmed these findings: they found that showering for 10 min and bathing for 20 min consistently resulted in at least a twofold increase in median THM in blood. They also state that "It is well established that THM concentrations in water in residential water heaters are generally much higher than in tap water from the utility distribution system." (Nuckols et al. 2005, p. 869).

9.2.4 Contribution of DBPs to the Estrogenic Effects in Drinking Water

The potential health risks of endocrine disrupting chemicals (EDCs) have been of great concern since the mid-1990s (Itoh et al. 2011). Many epidemiological studies have been conducted to assess the relationship between "adverse reproductive and developmental outcomes" and "exposure to chlorinated drinking water" (Itoh et al. 2011). Some reviews of these studies (Zavaleta et al. 1999, IPCS (International Programme on Chemical Safety) 2000, Nieuwenhuijsen et al. 2000 and USEPA 2006) pointed out that THMs and other chlorinated DBPs can lead to adverse outcomes, such as spontaneous abortion, stillbirth, low birth weight, neurotoxicity, and birth defects. Currently, hundreds of compounds have been identified as suspected EDCs (Endocrine Disruptor Screening and Testing Advisory Committee (EDSTAC 1998), and "most research on EDCs focuses on these individual micro-pollutants" (Itoh et al. 2011). Since epidemiological studies have discovered the relationship between exposure to DBPs formed from NOMs and reproductive and developmental toxicity, in addition to suspected EDCs, it is important to measure the effects of chlorinated DBPs in drinking water and of raw water containing both micro-pollutants and NOMs on reproductive and developmental outcomes (Itoh et al. 2011). Moreover, the Endocrine Disruptor Screening and Testing Advisory Committee (EDSTAC) (1998) established by the USEPA also recommended that a mixture of DBPs be evaluated for their potential to cause endocrine disruption.

Figure 9.1a illustrates the components of water that induce estrogenic effects and how they are changed by chlorination (Itoh et al. 2009). First, NOMs have a weak estrogenic effect that increases after chlorination, and then the effect increases gradually over time, even in the absence of residual chlorine (Itoh et al. 2009). The findings from Fig. 9.1a, b illustrate the components of the estrogenic effect that comes from NOMs. The estrogenic substances formed after chlorination as part of the chlorinated DBPs and the "estrogenic effect intermediates" change into estrogenic substances over time, and hence the estrogenic effect increases over time after chlorination. Itoh et al. (2011) further suggested that "to decrease the estrogenic effects in drinking water, NOMs in addition to suspected EDCs should be removed before chlorination. Furthermore, it is important to assess the reproductive and developmental toxicity of mixtures of DBPs that originated from NOMs."

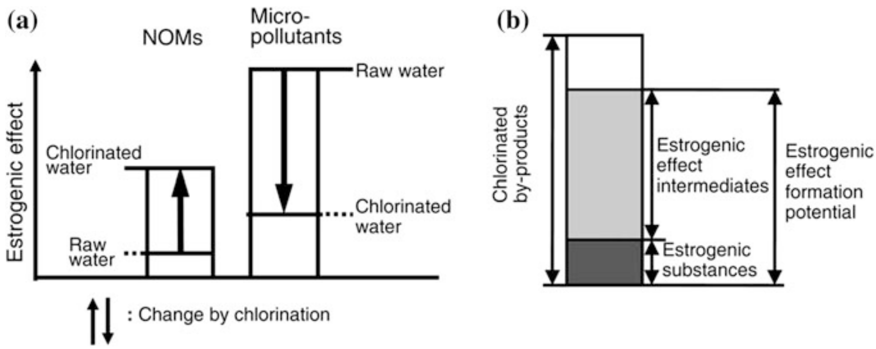

Fig. 9.1 Components of the estrogenic effects in chlorinated drinking water (Itoh et al. 2009)

9.3 Management Practices in Developed Countries

In USA and Canada, chlorine remains the most widely used method for disinfection of drinking water, while recently some alternative approaches such as chloramines have been applied to reduce the DBPs (Health Canada 2009 and Lenntech n.d 2014). The DBP rule to regulate DBPs has been developed by USEPA (USEPA 2003) in three stages. In stage 1, the total THMs standard and HAAs standard are 80 μg/L and 60 μg/L, respectively. A further reduction of standard for total THMs and HAAs to 40 μg/L and 30 μg/L, respectively, would occur under stage 2. To date, the USA is still in stage 1. Moreover, a review of the research is presented in Richardson et al. (2007); they examined 85 DBPs, of which 74 are as yet unregulated in the USA.[2] Richardson et al. conclude that brominated DBPs are in general more genotoxic and more carcinogenic than the chlorinated compounds. Iodinated DBPs are highly genotoxic, but their carcinogenic properties have not been tested. Richardson et al. further pointed out that the data is not complete even for the 11 DBPs which are currently regulated.

Adams et al. (2005) analyzed data on drinking water treatment systems in the State of Missouri for the years 1997–2001. Their results show that regulatory limits for THMs and HAAs were exceeded during this period by a significant number of treatment plants, mainly those serving populations of fewer than 10,000. Recently, USEPA has conducted a comprehensive toxicological evaluation of DBP concentrates because the widespread use of chlorine in untreated water in the USA results in the formation of DBPs (Pressman et al. 2010). The research project is titled "Integrated Disinfection Byproducts Mixtures Research: Toxicological and Chemical Evaluation of Alternative Disinfection Treatment Scenarios", also known as the "Four Lab Study" since the multidisciplinary team of researchers come from

[2] Currently, 11 DBPs are regulated by USEPA, including four THMs, five HAAs, Bromate, and Chlorite.

four national Laboratories or Centers of the US Environmental Protection Agency's (EPA's) Office of Research and Development. The purpose of the project is to develop (a) a new procedure for producing chlorinated drinking water concentrate for animal toxicology experiments, (b) comprehensive identification of at least 100 DBPs, and (c) quantification of 75 of the priority and regulated DBPs (Pressman et al. 2010).

The results from studies on risks to human health by the use of chlorine reviewed above seem to suggest that the Health Canada guideline for total THMs of 100 μg/L (Health Canada 2006) and of 80 μg/L (Health Canada 2008) for HAAs is out of date. Even the reduction of MAC for THMs to 80 μg/L in Ontario may be unsafe. For some Ontario municipalities, the total THMs far exceed the regulatory limit, with the average of the 90th percentile being 93.8. The 95th and 99th percentile values for Ontario are 106.02 and 152.88, respectively.

In 2009, Health Canada issued a national consultation document on chlorine in drinking water (Health Canada 2009). Its primary concern was with disinfection, and while Health Canada brought in a limit for BDCM of a maximum of 16 μg/L (Health Canada 2006), the maximum limit of THMs remained unchanged (at 100 μg/L), But Health Canada 2009 states that: "Disinfection is essential to safeguard drinking water; the health risks from disinfection byproducts are much less than the risks from consuming water that has not been disinfected" (Health Canada 2009, p.1) This is largely a "benefit-cost" conclusion rather than a serious assessment of risks. In fact the document states that the Guideline… "does not review the benefits or the processes of chlorination, nor does it assess the health risks related to exposure to byproducts formed as a result of the chlorination process." How can a "Health Canada" guideline fail to assess the health risks…of exposure to disinfection byproducts? The document goes on to state: "Health Canada has classified chlorine as unlikely to be carcinogenic to humans. Studies in laboratory animals and humans indicate that chlorine exhibits low toxicity, regardless of the route of exposure (i.e. ingestion, inhalation, dermal). Studies in animals have not been able to identify a concentration of chlorine associated with adverse health effects, in part because of aversion to its taste and odor. No adverse health effects have been observed in humans from consuming water with high chlorine levels (up to 50 mg/L) over a short period of time." It supports a free chlorine residual of 200 μg/L in the distribution system to prevent regrowth of bacteria. It concludes boldly that: "Because chlorine is not stable under environmental conditions, exposure is not expected to be significant, and there are few data available" (Health Canada 2009, p.16). It contains the following statement: "[T]here have not been any epidemiological studies that have specifically examined free chlorine concentrations in water and long-term health effects in the human population." This assertion is completely out of date, as shown above; the study by Hwang et al. (2008) raises important questions and suggests that any level greater that 4 μg/L carries serious risk for the nursing mothers.

In contrast, some developed countries in the EU have applied alternative approaches for drinking water disinfection to minimize the use of chlorine. For example, France and Italy use Ozone as a primary disinfectant. In Germany, the drinking water treatment companies commonly use Ozone for drinking water disinfection, while chlorine or chlorine dioxide is used only if it is required. An engineer from a drinking water treatment company in Hamburg, Germany said that they have not used chlorine to disinfect their distribution system since the beginning of the 1950s and there is no chlorine residual at all in their 10 waterworks and the distribution system.[3]

In Denmark there has been a policy of gradual elimination of *all* chlorine from their water treatment plants. In fact, according to the online edition of Copenhagen Post (2009, June 3), Copenhagen became the last municipality to rely completely on underground aquifers and completely stopped using all chlorine after using it for the past 37 years. They have no need to worry about THMs, as there are none in their drinking water.

In the Netherlands, they have gone considerably further in that as of 2005, *no chlorine is used at all* (Smeets et al. 2009). From 1976 onward, the use of chlorine has been steadily reduced until 2005, when the last use of chlorine was replaced by UV. Moreover, according to Smeets et al. (2009, p. 3), "UV inactivates a wider spectrum of pathogens than chemical disinfection, and microbial safety is easily warranted by process monitoring and control." Note also that no chlorine is used in the distribution system; the approach is to "starve" regrowth of pathogens rather than rely on disinfection. To quote again:

> There was no more need for a disinfectant residual during distribution to prevent regrowth. The level of post-disinfection at surface water treatment plants was lowered to such an extent that, in 2008, no chlorine is being applied at all, and the few locations where chemical disinfection is applied (chlorine dioxide) no residual disinfectant can be measured in the distributed water." Thus, the Netherlands has more or less completely eliminated THMs and HAAs.

9.4 Conclusion

Recently epidemiological studies have confirmed associations between human health effects and exposure to chlorinated DBPs. The evidence for carcinogenicity of DBPs is strongest for bladder cancer, while some but not all findings have reported positive associations between colon and rectal cancer and DBP exposure. In addition, some epidemiological studies also reported associations between consumption of chlorinated water and adverse reproductive outcomes, including preterm births and defects in the unborn child. The regulation of DBPs has played an important role for safe drinking water and public health; however, more than 50 percent of the toxic halides formed during disinfection have not been defined. In some developed countries, particularly in EU countries, alternative methods of disinfection of drinking water such as Ozone and UV and cartridge filtration are

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[3] Personal communication by E-mail, from Dr M. Scheemann, Hamburg.

being used to minimize the use of chlorine. But in the USA and Canada, chlorine remains the most widely used method of disinfection of drinking water. Therefore, it seems clear that (1) comprehensive toxicological evaluation of whole DBP mixtures are necessary, and (2) greater emphasis must be placed on continuing to reduce the allowable concentrations of all toxic halides in drinking water. As a long-term policy, it would be sensible to follow the example of the European countries that have completely eliminated the use of chlorine in drinking water.

In the past, the use of chlorine has been shown to have benefitted large populations all over the world. For example, typhoid fever had killed about 25 out of 100,000 people in the US annually, a death rate close to that now associated with automobile accidents. Today, typhoid fever has been virtually eliminated. But the new evidence suggests grave long-term health risks associated with the use of chlorine. Chapter 3 contains a review of drinking water treatment technologies, which clearly shows that there are alternatives for disinfection that are cost-effective. Therefore, we can conclude that chlorination of drinking water is now an obsolete technology, and it is high time that North America moved away from chlorination and followed the example of the Netherlands, Denmark, and Germany.

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Chapter 10
Public Health and Lead Sampling Protocols for Drinking Water: A Critical Review

10.1 Introduction

The objective of this chapter is to review critically lead sampling protocols for drinking water from the point of view of public health. There are several lead sampling protocols that can be used. In the US, the EPA requires that a 1 L sample be taken after drinking water has been stagnant in the drinking water pipe for at least 6 h in a house or any other location from which drinking water is drawn; we refer to this as a period of 6 h stagnation. In Europe, members of the European Union are guided by the Drinking Water Directive (DWD), which does not spell out the sampling protocol, but in practice EU countries rely on a 1999 major report called "Developing a new protocol for the monitoring of lead in drinking water" by authors Van den Hoven et al. (1999). This was published as a European Commission Report No. 19087 in 1999. In this chapter, we refer to this major report simply as the "EU Report."

In Canada there appear to be two approaches. The Federal Government proposed a set of guidelines for discussion in 2007. After a review and comment period, the Canadian Federal Government released a newly revised set of Guidelines that accommodates the fact that Ontario has adopted its own set of regulations governing lead sampling which are closer to the practice of the European Union. In Australia, the Australian Drinking Water Guidelines (ADWG) are not legally binding (National Health and Medical Research Council (NHMRC) 2004a). Samples used to test for lead are taken outside the consumer's property from a service pipeline by water authorities, and are supposed to be carried out on a monthly basis. According to the ADWG, sampling at the consumer's tap can take place when special cases of leaching of metal and other corrosivity-related issues are suspected (NHMRC 2004b). However, no common standard sampling protocol is defined in the ADWG.

This chapter reviews the scientific evidence that forms the basis for the sampling protocols used by the USEPA and the European Union. In our opinion, serious

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policy errors have been made in Europe and in Canada in adopting sampling protocols that are not consistent with the scientific best practice. The chapter is organized as follows. Section 10.2 reviews a select set of research on blood lead levels (BLLs) and their potential health effects, including the associated social costs. Section 10.3 reviews the nonbinding Canadian Federal Guidelines and the very different lead sampling protocol adopted in Ontario, Canada. Section 10.4 is a critique of the European report that is the basis of the European lead sampling protocol; it is shown that the 30 min sampling protocol used in Europe (and in Ontario) is not supported by the report on the scientific work carried out in Europe by van Hoven et al. (1999) for the European Commission; it is also inconsistent with the science of lead leaching into water. Section 10.5 restates the protocol adopted by the USEPA, which we believe is consistent with science. Some policy lessons are presented in the conclusion.

10.2 Adverse Health Risks and Social Costs Associated with Lead in Drinking Water

There is a large literature on the evidence of lead in blood as a result of lead found in drinking water. In this section, we review work published since the early 1980s in North America and Europe, dealing with exposure to lead from drinking water and from other sources. We organize this review into three subsections: one on the amount of lead found in blood samples, the second on the adverse health effects of lead in blood, and the third on the social costs associated with these adverse health effects.

10.2.1 Amount of Lead in Blood

The health risks associated with exposure to lead are well documented, having been the subject of many studies over a long period. Early studies showed a reduction in blood lead levels can result from the hardening of soft water supplies (Gallacher et al. 1983), and increasing water pH level as well as replacing lead pipes (Sherlock et al. 1984). The removal of lead pipes can result in a decline in the median BLL from 26 to 13 μg/100 ml, while the reduction in plumbosolvency by raising pH can result in a decline of median blood lead from 21 to 13 μg/100 ml (Sherlock et al. 1984). Elwood et al. (1984) also found a significant contribution to BLLs from drinking water. Thus almost all the lead in drinking water results from the use of lead pipes and lead fixtures and meters, which also contain lead. Although new buildings are required to use “lead free” pipes, lead in drinking water continues to be a problem in many regions. “Lead free” pipes according to the USEPA mean that

Table 10.1 Summary of findings on blood lead levels in drinking water in publications dating before 2000

Author/year	Range of lead ingested by		Contribution to BLL
	Adults	Children	
Elwood et al. (1984)	1 mg		4.9 and 5.5 μg/dL
Lacey et al. (1985)		1 mg/L	0.62 mg/L
Bois et al. (1989)	10 μg/L		Up to 7 percent of BLLs
Houk et al. (1989)	50 μg/L		Up to 22 percent of BLLs
Health Canada (1992)		2.9 μg per day of ingested lead	1.45 μg are absorbed (50 percent)
Health Canada (1992)	7.2 μg of lead ingested per day		0.72 μg are absorbed (10 percent)

pipes, pipe fittings, and pumps contain no more than 8 percent lead while solder and flux should not contain more than 0.2 percent lead (USEPA 2010). The earliest reference we found that attempts to measure lead in blood was probably by Sayre et al. (1974). Since then there has been a large literature on the contribution of lead to BLL. The findings published prior to 2000 are summarized in Table 10.1.

The results in Table 10.1 show an absorption rate of approximately 50 percent for children under 2 years of age while for adults it can be as high as 22 percent. This is consistent with findings in many studies that children are more at risk than adults. According to Moore et al. (1985), children are more at risk because as lead in water is in soluble form, it is more readily absorbed into the bloodstream than food containing lead; lead in food tends to combine with other elements to form compounds that may not be easily digestible. Lacey et al. (1985) also support this finding. They carried out a study in Glasgow on the effects of lead on BLLs in the diet of 10–12-week-old infants and showed that although lead from baby food was detectable, it was an almost insignificant amount when compared with lead ingested from drinking water. Dust, which is a large contributor to total lead ingestion in adults, has been shown to contribute insignificantly to BLLs in a statistical study conducted by Elwood et al. (1984).

The relationship between lead in water and BLLs is nonlinear (Sherlock et al. 1984 and Moore et al. 1985). This means that for every unit decrease in lead, a more than proportionate decline in blood lead level can be achieved (Moore et al. 1985). Other studies also examine lead exposure from drinking water and other sources either in terms of its relative contribution (i.e. percentage of total exposure) or its absolute contribution (i.e. exposure measured in units such as micrograms per liter). In a study conducted in 1988 in Ontario (Canada) by Graham (1988), a composite sample was used in 40 homes to determine the average amount of lead a typical family consumes in 1 week. The results showed that the average concentration of lead was between 1.1 and 30.7 μg/L with a median of 4.8 μg/L. According to Health Canada (1992), Graham's findings translate to an ingestion of lead of 7.2 μg and 2.9 μg per day for an adult and a 2-year-old child, respectively, if we assume that an adult consumes 1.5 L and a child consumes 0.6 L of drinking water per day.

The same Health Canada (1992) study showed that (1) drinking water accounts for 9.8 and 11.3 percent of total lead intake for a child and an adult respectively; (2) for the remaining sources of lead exposure food, air and dust account for 50.9, 1.2, and 38 percent respectively for 2-year-old children and 78.0, 7.1, and 4.2 percent respectively for adults; (3) of the total lead that is absorbed into the bloodstream, drinking water accounts for 10.7 percent of total lead for adults and 11.6 percent for a 2-year-old child; (4) of the 2.9 μg of lead ingested per day by a 2-year old, 1.45 μg is absorbed into the bloodstream, while only 0.72 μg of the 7.2 μg of lead ingested per day by adults is absorbed; and (5) compared to other sources of lead exposure, food and water have the highest absorption rate (50 percent) for 2-year-old children. For children under 2 years of age, in particular bottle-fed infants, the absorption rate from water used for dietetic purposes is much higher.

In more recent work, Fertmann et al. (2004) found significant correlation between BLLs and average lead concentration in tap water after examining BLLs in over 200 women in Hamburg. They also found that women from households that had drinking water lead above 5 μg /L had significantly higher BLLs than those that had no detectable lead in their drinking water. A follow-up study on a subset of the group revealed a significant reduction in BLLs after they were advised either to flush prior to consuming water or to consume bottled water. Edwards et al. (2009) found strong correlation between lead levels in drinking water and BLLs above 10 μg/dL for children under 1.3 years of age. Surprisingly, the correlation between BLL and drinking water lead was weaker when children up to 2.5 years of age were included in the study.

The above studies indicate a good correlation between BLLs and drinking water at the tap. However, changes to water treatment processes prior to distribution to residential homes can also influence blood lead levels. Silicofluorides, fluosilicic acid, and sodium fluosilicate are used to fluoridate over 90 percent of US municipal water, but have been shown to increase the risk of elevated blood lead (i.e. BLLs over 10 μg/L) in children by up to 200 percent when compared to children consuming non-fluoridated water (Coplan et al. 2007). Miranda et al. (2007) conducted a study in Wayne County, North Carolina, which showed that when municipal water authorities switched their disinfection treatment process to chloramine disinfection, it might have led to increased BLLs. Levin et al. (2008) reviewed the contribution to BLLs from various sources of lead exposure and found that changes in water treatment processes by water authorities can influence BLLs, and that the risk of elevated BLLs in children is greater when their homes have lead service lines.

10.2.2 Health Effects of Lead in Blood

Fowler and Duval (1991) studied the effects of lead on kidneys and found that short-term high-dose lead consumed from drinking water can induce nephropathy, a condition that can result in a reduction in the organs' ability to filter toxins and protein effectively. In two earlier studies, Cullen et al. (1984) and Wildt et al. (1983) had found reduced sperm count levels in men with BLLs of up to 50 μg/L. Adverse health effects even from low BLLs have also been found. Decreased height and delayed breast development in pubescent females have been associated with BLLs of 3 μg/dL (Selevan et al. 2003). (Note that a deciliter (dL) is one-tenth of a liter.) Meta analysis studies by Schwartz (1994) and Levin et al. (2008) indicated no sound evidence of threshold effects for BLLs.

Some studies have addressed the serious risks to cognitive and intellectual development to which children may be exposed as a result of lead levels in water. McMichael et al. (1988) discovered reduced IQ and cognitive development index in children with BLLs of less than 6 μg/L and have shown that newborn children with BLLs of less than 3 μg/L had higher scores in their cognitive development index tests than those born with lead levels of over 6 μg/L. Moreover, BLLs of 10 μg/dL in 9-year-old children have been associated with higher dropout rates and increased criminal activities at a later age than children who had BLLs of 5 μg/dL (Needleman et al. 1990).

Later studies have confirmed these findings. Lanphear et al. (2005) found intellectual deficits for children who had BLLs less than 7.5 μg/L, while Lanphear et al. (2000) found an inverse relationship between mathematics and reading scores and BLLs in children aged 6–16. In the same study, Lanphear et al. (2000) found that an increase in BLL by 1 μg/dL can cause a 0.7 and 1.0 point decline in mean arithmetic and reading scores respectively. Canfield et al. (2003) found that an increase in lifetime BLLs from 1 to 10 μg/dL can reduce IQ by 7.4 points, while Lanphear et al. (2005) showed a reduction of 6.2 IQ points (with a 95 percent confidence interval between 3.8 and 8.6) for the same BLL range as the Canfield study; a log-linear model confirmed that IQ levels declined the most when BLLs were less than 10 μg/L and indicated a nonlinear relationship between BLLs and IQ. Indeed, the results indicate that an increase in BLLs from 2.4 to 10 μg/L can be associated with a drop in IQ by 3.9 points, while an increase in BLLs from 10 to 20 and 20 to 30 μg/L can be associated with a drop in IQ levels by 1.9 and 1.1 points, respectively (Lanphear et al. 2005). Zahran et al. (2009) examined the effects of BLLs on fourth grade students' subject scores including mathematics, science, and language. Regression analysis indicated student performance on a wide range of subjects declines significantly when BLLs exceed 10 μg/dL. Lead ingestion in the formative years of life can have long-lasting effects later on in life. Cognition and neurobehavioral patterns can be affected negatively as a result of even low-level lead exposure during early childhood (Brubaker et al. 2009).

Lead in drinking water can account for at least 15 percent of total lead when an individual is exposed to lead on a daily basis according to Bois et al. (1989). Health

Canada (1992) indicates a 10.7 percent contribution to lead exposure from drinking water in Canadian adults, while the USEPA (1993) estimates the exposure to the general population of the US to be between 10 to 20 percent. Regardless of the relative exposure, drinking water is still one of the largest *controllable* sources of lead exposure. Lanphear et al. (1998) suggested that leaded water contributed significantly toward BLLs in children after adjusting for other sources of lead exposure. Young children are particularly affected due to the ease of absorption of lead into the bloodstream. Children at 24 months can have a lead uptake of up to 50 percent of the lead they ingest (Health Canada 1992 and Mushak 1998).

These findings indicate that children as well as adults can be exposed to serious risks to their intellectual and cognitive development at lead levels in drinking water that are easily achievable in many homes in Canada and the US under the current Maximum Contamination Levels (MCL). Although many steps have been taken over the years to reduce the amount of lead in water, dangerously high levels of lead exposure in drinking water can occur in major cities. In 2003, many homes in Washington DC were found to have lead levels of over 15 µg/L. This prompted the District of Columbia Water and Sewer Authority to take action by replacing lead service lines, installing water filters in homes, and adding phosphoric acid to reduce the corrosivity of the water. According to Maas et al. (2005), since the 2003 event, lead above 10 µg/L can still be found in 15 percent of homes in DC when the 6 h stagnation protocol is used to measure lead. An independent testing conducted in 2009 at six public schools in Washington DC found elevated lead levels in up to 41 percent of drinking sources (Triantafyllidou et al. 2009). Washington DC was not the only region to experience elevated lead levels in drinking water in public schools. An examination of drinking water lead levels in 292 public schools in Philadelphia indicated over 57 percent (168) of schools had lead levels over 20 µg/L (Bryant 2004). Even more alarming was that 34 schools had lead levels between 50 and 100 µg/L and a further 50 schools had lead levels above 100 µg /L.

10.2.3 Social Costs of Lead in Drinking Water

Grosse et al. (2002) used data from 1976 onwards to calculate changes in worker productivity related to BLL. Their findings can be summarized as follows:

(1) While using the assumption that IQ points decrease by 0.185–0.323 for every 1 µg of lead per dL, each IQ point raises worker productivity by 1.76–2.38 percent;
(2) The value of one IQ point in 2000 dollars lies within the range of $12,700 and $17,200;
(3) The economic gain from reduction of lead levels for each individual is $29,000–$83,800, while
(4) The overall gain to society can lie between $110 and $318 billion (in constant 2000 dollars).

Gould (2009) conducted a cost–benefit analysis of social and economic benefits that can arise from lead reduction in household paints and found:

(1) The cost of lead paint hazard control ranged from $1 to $11 billion;
(2) The monetary gain as a result of lead hazard control amounted to $11–$53 billion for reduction in medical treatment costs, $165 to $223 billion for increased earnings potential, $25 to $35 billion for increased tax revenue, $30 to $146 million in reduced special education costs, $267 million for reduction in treatment of lead-linked ADHD cases, and a gain of $1.7 billion stemming from reduced criminal activity linked to lead; and
(3) The net benefit to society can be worth as much as $181 to $269 billion, a return of $17–$221 per each dollar invested in hazard control.

Landrigan et al. (2002) estimated the costs of four categories of illnesses: lead poisoning, asthma, cancer, and neurobehavioral disorders. They identified the main consequence of lead poisoning as the loss of IQ over one's lifetime. The estimated loss in lifetime earnings for a 1 year cohort of 5-year-old boys in 1997 was $27.8 billion, while it was $15.6 billion for girls of the same year cohort (BLLs were relatively the same for both groups). The total loss to society was estimated at $43.4 billion for lead poisoning, while the total annual costs of environmentally attributable diseases ranged from $48.8 to 64.8 billion (Landrigan et al. 2002). A similar study conducted by Davies (2006) in Washington State estimated the loss in lifetime earnings for 5-year-old boys at $947.4 and $531.5 million for 5-year-old girls for a total of $1478.8 million in 2004 dollars. Stefanak et al. (2005) estimated the cost of child lead poisoning in Mahoning County, Ohio, on the healthcare system. They found that the cost of screening and treating each child for lead poisoning increases as BLLs increase. Children with BLLs over 20 μg/dL on average can cost the system $969 per child compared to $29 for children with BLLs under 10 μg/L; the total cost in 2002 was $124,653. Stefanak et al. (2005) also estimated the cost associated with the effects of lead on juvenile delinquency and special education. The (discounted) cost for juvenile justice services for children with BLLs greater than 25 μg/dL is $223,536 for each 1 year cohort of children. Special education cost was estimated at $85,295 for each 1 year cohort of children and the total cost to the system was estimated at $499,484. Zahran et al. (2009) suggested that a one-time payment for preschool lead exposure prevention would be more cost-effective than having to pay periodically for the future costs associated with neurotoxic damage associated with lead exposure during preschool years.

10.3 The Canadian Federal Guidelines for a Protocol for Sampling Drinking Water

10.3.1 Stagnation Time and Sampling Protocols

Lead concentrations in drinking water can be largely due to the length of time for which water dwells in a plumbing system before use for dietetic purposes. Also, the *volume* of water (if any) that is not used prior to its use for dietetic purposes plays a crucial role in human exposure to lead from drinking water. The time taken between uses for dietetic purposes or inter-use stagnation time is also important and can result in large variations in lead concentrations (see results of EU Report 1999 and Bailey et al. 1986a). As a result the proper sampling protocol used to determine the amount of lead in drinking water is crucial in minimizing the health risks and social costs associated with lead in drinking water.

In Canada, the Federal government is responsible for drinking water standards on federal lands, in areas where the Federal government is the water supply owner and in areas that fall under federal jurisdiction, e.g. First Nations lands, national parks, and on-board common carriers (ships, airplanes, etc.) (Canadian Council of Ministers of the Environment (CCME) 2004). The *Constitution Act* of 1867 gave ownership of surface and groundwater to the provinces, and provincial governments have legislative responsibility for providing safe drinking water from those sources. Municipalities obtain power from the provincial level in order to pass by-laws that can also impact water resources. The three territories (Northwest Territories, Nunavut, and Yukon) do not have ownership of their natural resources, including water, but are still responsible for the provision (and legislation) of safe drinking water (CCME 2004).

10.3.2 Canadian Federal Guidelines for Lead Sampling Protocols

There appear to be two options for monitoring lead at residential sites under the new Federal guidelines. Option 1 is a "two-tier" approach for assessing corrosion control in a distribution system. With this option the first tier requires a 1 L sample to be taken after a period of at least 6 h stagnation. The sample is to be taken at the kitchen tap or the source where drinking water is most commonly taken. If 10 percent of the sites have lead concentrations above 15 μg/L, then the following corrective actions are recommended:

(1) Initiation of a public education program which includes encouraging consumers to flush water after prolonged stagnation.
(2) Conducting additional sampling from at least 10 percent of the sites with the highest lead concentrations (above 15 μg/L).

(3) Informing consumers of test results and corrective measures to reduce lead exposure. This includes flushing of the plumbing system before use for dietetic purposes, replacing leaded fittings and fixtures, replacing lead service lines, and using water treatment devices.
(4) Implementing corrosion control measures within the distribution system. These measures can include the adjustment of pH and alkalinity, addition of corrosion inhibitors, and replacing of lead service lines.
(5) Encouraging homeowners to clean debris from aerators and screens (since these are not required to be taken off before taking the 1 L sample).

Tier 2 of the sampling protocol is taken when more than 10 percent of the sites tested under the Tier 1 sampling protocol exceed 15 μg/L. Under Tier 2, four consecutive 1 L samples are taken from the tap after a period of at least 6 h stagnation. Each 1 L sample is analyzed individually and a stagnation profile built.

Option 2 of the new Federal Guidelines is intended for "…jurisdictions in which sampling after a 6 h stagnation time is not practical or regulatory obligations restrict the use of the two-tier approach…" (Health Canada 2009). For this Option, four consecutive 1 L samples are to be taken at the tap after the tap is *flushed for 5 min and left to stagnate for 30 min*. Option 2 was intended to evaluate corrosion at properties that have lead service lines and was not intended for system wide evaluation of corrosion or corrosion control optimization (Health Canada 2009). If average lead concentration from the four samples is greater than 10 μg/L in more than 10 percent of the sites monitored, then additional corrective measures are to be taken similar to the 5 steps taken in Option 1 above. These measures also include resorting to the Tier 2 sampling protocol identified in Option1 to assess fully and to remedy properly the corrosion problem. The sampling frequency and selection of sites for residential monitoring is exactly the same as that of the EPA protocol (see Table 10.2).

Table 10.2 Suggested number of monitoring sites (adapted from USEPA 2000, as cited in Health Canada 2007)

System size (number of people served)	Number of sites (initial monitoring: once per year)	Number of sites (reduced monitoring: once per year)
>100,000	100	50
10,001–100,000	60	30
3,301–10,000	40	20
501–3,300	20	10
101–500	10	5
Less than or equal to 100	5	5

10.3.3 The Ontario Lead Sampling Protocol

The Ontario provincial regulatory requirements for safe drinking water are in many ways different from the Federal guidelines and are largely based on the 1999 EU Report. Ontario Regulation 170/03 under the Safe Drinking Water Act of 2002 sets out clearly defined requirements in Schedules 15.1 and 15.2 for sampling protocols that must be used to determine lead in drinking water in municipal and nonmunicipal buildings (both residential and nonresidential types). For large and small municipal residential and nonmunicipal year-round residential properties, the Regulation requires three samples to be taken at the kitchen tap or the tap that is most commonly used for drinking water purposes. The first sample is to be taken after a stagnation time of no less than 30 min but no more than 35 min. This stagnation period will commence after a period of 5 min of flushing at the tap. The second sample is to be taken immediately after the first sample without turning off the tap or altering the flow rate of water. A third sample is to be taken immediately after the second sample without turning off the tap or altering the flow rate of water. The first two samples are to be tested for lead, while the third sample is tested for pH. Apart from obtaining samples from the tap (source for drinking purposes), three samples have to be taken from a point in the distribution system. From the distribution point, water is to be flushed before the three samples are taken until the quality of water is representative of water in that part of the distribution system. The first sample from the distribution system is tested for lead while the second and third samples are tested for alkalinity and pH respectively. For large and small municipal nonresidential buildings and for seasonal, large, and small nonmunicipal buildings only one sample is required to be taken annually from a point in the distribution system or a point in a plumbing system which is suspected to have elevated lead concentrations·

Samples should be taken every 12 months during the following times: (1) between December 15 and April 15 and (2) between June 15 and October 15. The number of samples taken from both plumbing and distribution systems would be determined by the number of people served by a particular drinking water system (Table 10.3 shows the number of samples per location).

The stagnation time under the Ontario Safe Drinking Water Act 2002 is based on a report of the Ontario Drinking Water Advisory Committee, which made recommendations to the Ontario government. The Drinking Water Advisory Committee's report is primarily based on the 1999 EU Report, mentioned above.

10.3.4 The 1999 EU Report

The 1999 EU Report evaluated the performance of several sampling protocols including:

(1) Random Day Time (RDT): A sample (usually 1 l) taken randomly during normal working hours from a drinking water tap without any prior flushing.

Table 10.3 Standard number of sampling locations (Government of Ontario 2010)

Column 1	Column 2	Column 3	Column 4	Column 5
Item	Population served by drinking water system	Number of sampling points in plumbing that serves private residences	Number of sampling points in plumbing that does not serve private residences	Number of sampling points in distribution system
1.	1–99	5	1	1
2.	100–499	10	1	2
3.	500–3,299	20	2	4
4.	3,300–9,999	40	4	8
5.	10,000–49,999	60	6	12
6.	50,000–99,999	80	8	16
7.	100,000 or more	100	10	20

(2) Fully Flushed (FF): A sample taken after a period of flushing at the drinking water tap.
(3) 30 Minute Stagnation (30MS): After a period of flushing at the tap, water is allowed to remain in the system for 30 min before a sample is taken.

These sampling protocols were evaluated against the *composite proportional sampling (COMP) method*. The COMP sampler is a sampling device, which is attached to the consumer's kitchen tap in order to determine the average lead concentration over a period of 1 week. It is a consumer-operated device which, when turned on properly, captures 5 percent of volume of water drawn. Consumers are required to turn on the device only when they are consuming water for dietetic purposes. According to the EU Report, the COMP sample is the only method that *captures all the factors influencing average weekly lead intake by consumers* (EU Report, p. 32).

The EU Report recommended that either the RDT or the 30MS be used as protocols for statutory monitoring purposes and zone assessment while the effect of treating water, e.g. orthophosphate dosing at a treatment plant, can be assessed using RDT and lead pipe test (which involves setting up a lead pipe rig at treatment facilities and having samples taken after a 24 h stagnation period). For an "accurate, and repeatable, value for average weekly lead concentration" (EU Report, 9. iv), the Report recommends the *30MS sampling protocol.* The stagnation time of 30 min was based on findings from Bailey et al. (1986b), Baron (1996) and Van den Hoven (1986), which showed strong correlation between COMP and 30MS times. Note that this is a statistical correlation, without any basis in metal chemistry. The following section provides a critique of the EU's assessment of the various sampling protocols. We argue that in the case of statutory monitoring, an average of FF and RDT provides a better estimate than the 30MS protocol, while providing lower cost and better consumer acceptance. *These findings are based on the EU Report itself.* Indeed the critique will show that the RDT by itself can provide as good an estimate as the 30MS protocol and would therefore be less costly to use.

10.4 A Critique of the EU 1999 Report

The aim of the EU Report was to assess the performance of several sampling protocols, which are used to test lead in drinking water, based on several criteria: reproducibility, practicality, cost, consumer acceptance, and representativeness. The EU Report attempted to address the difficulties that arise in establishing a proper sampling method. Over the years the definition of a 'representative sample' in the European DWD has changed for monitoring purposes. According to the EU Report, this definition changed from "…a sample of water intended for human consumption obtained from a proportional flow device at the tap…" in 1995 to "sample of water obtained by an adequate sampling method at the tap and taken so as to be representative of the weekly average value ingested by consumers" in 1997 (EU Report, p. 14). One of the stated objectives, therefore, was to "develop a monitoring system for lead to be assessed on the basis of a sample that is 'representative of that consumed by man'" (EU Report, p. 5). As stated before, three sampling protocols were assessed in the report:

(1) 30MS time,
(2) RDT, and
(3) FF sample.

The 30MS protocol was further broken down into the 30MS1, 30MS2, and 30MSA, which stand for the first liter taken after 30MS, second liter taken after 30MS, and the average of the first and second liter, respectively. The average of the RDT and FF was also used for further comparisons. The lead per liter for each sample obtained in each sampling protocol was compared to the amount of lead per liter in the COMP sampler because the latter was taken to be the real or "true value" of the average weekly intake of water by consumers. Eleven test areas (with approximately 30 samples from each area) were selected to be "representative of all combinations of water types and plumbing materials found in Member States" (EU Report, p. i). This sample, however, is not representative of the true population of Member States since at least 50 percent of the houses sampled in each test area were required to have lead plumbing (EU Report, p. 27). Therefore, sampling for each test area was not truly random.

"Representativeness" for individual properties was determined by the ratio of test procedure and reference method (COMP) while "representativeness" for the supply area was determined by the slope and correlation coefficient of the linear relation between each procedure and COMP as well as the average value of tested protocol compared to the average COMP value in the distribution area. "Reproducibility" of the sampling protocols was determined by the coefficient of variation or relative range of the three samples taken in one property. The ability of the sampling protocols to detect "problem properties" was assessed by analyzing the proportion of properties where the attached lead measuring device (COMP) exceeded the MCL of 10 μg/L; "false positives" were also analyzed, i.e. the proportion of a sampling protocol that were greater than 10 μg/L when the measured lead was in fact less than 10 μg/L. "Consumer acceptance" was assessed by the consumer's willingness to cooperate in undertaking a sampling protocol in their home while "practicality" was assessed by "several aspects of the procedure (e.g. is the procedure easily applicable, are skilled samplers needed, does the procedure need specific tools, etc...)" (EU Report, p. 56). Both "consumer acceptance" and "practicality"' have no quantifiable evaluation schemes. "Cost" was assessed using a hypothetical wage rate, and an estimated time in each sampling protocol was used to determine the total cost for each protocol.

In the EU Report's performance evaluation of all the sampling protocols, the 30MS and the RDT methods met the representativeness criteria while the FF sample did not. One of the ways in which representativeness was assessed was by examining the linear relationship between lead from a sampling protocol and lead from the sample obtained from the lead measuring device (called COMP). Both slope and R-squared statistic were used to make judgments. From the EU Report (see p. 35), the RDT sample overestimated the measuring device sample (slope 1.27 and r-squared of 0.61) while the 30 min first liter sample and the second sample both underestimated the COMP sample (slope of 0.80 and r-squared of 0.50 and 0.56 respectively). The average of the 2 L from the 30MSA, however, has a similar outcome for slope of 0.80 but a slightly improved r-squared of 0.58. FF strongly underestimated the measure of lead from the measuring device (COMP) with a slope of 0.57 with an r-squared of 0.29. If one were to choose a sampling protocol based on model fit, then *the RDT protocol should have been chosen.* If one were to choose a sampling method based on accuracy then the 30MS should be chosen but caution should be applied. Since the slope of the linear relationship between measured lead (COMP) and 30MS sample is 0.80, the 30MS sampling protocol consistently underestimates lead in drinking water. Therefore, on average, the 30MS sampling protocol underestimates the true value of lead by 20 percent. This has severe implications for determining the percentage or number of households with lead above the MCL of 10 μg/L, as the number of properties that actually have lead in drinking water is much more than expected under this protocol. One way to overcome this problem is to lower the MCL of 10 μg/L by 20 percent (i.e. to 8 μg/L). The RDT sampling protocol has the opposite problem. On average it overestimates the true value of lead by 27 percent.

The EU Report also did not consider the average of the RDT and the FF sampling protocol to be any better than the 30MS sample or RDT sampling protocols, stating that it "does not improve relation or give additional information" (EU Report, p. 38). However, the average of the RDT and the FF sample does in fact give a better (and an overall best) r-squared of 0.63 which was erroneously stated in the EU Report (see p. 38) as 0.58. The average of the RDT and FF sample also has a slope of 0.92 (for linear correlation with measured lead (COMP) sample); although the average of the RDT and FF underestimates the COMP sample, it still provides the best estimate (slope closest to 1.0 in relation to COMP) of all the protocols. The average of RDT and FF provides the most representative sample when compared with other protocols, which is not very surprising since RDT takes into consideration many inter-use stagnation times (although the stagnation times themselves are not known) in much the same way the COMP sample does. FF samples consider another dimension of consumer behavior; drinking water for dietetic purposes after running water for several minutes, e.g. drinking water or using water to cook after washing dishes. Although FF samples are not likely to be representative of consumer behavior, they still capture some element of it. Together, RDT and FF samples capture more elements of consumer behavior than a single sampling protocol alone would (i.e. RDT and FF together would capture more elements of consumer behavior than just the 30MS alone). The EU Report's overall evaluation of the RDT sampling protocol was that it was "unexpectedly good" which according to the Report can be explained by the fact that in general RDT overestimates the average weekly intake of lead in drinking water (EU Report, p. 65). The EU Report also further explained that the RDT sample was capturing some elements of the water consumption behavior of the consumer which was close to or greater than the average interuse stagnation time. The overestimation of the RDT compared to the COMP can be due to other factors as well. We believe that there is a possibility that the RDT overestimates the COMP sample because of the time of day in which the RDT samples were taken. The RDT samples were taken "during office hours, avoiding the periods of frequent water use (breakfast, lunch, and dinner) and the period of overnight stagnation" (EU Report, p. 20). In other words, the RDT sample was taken when there is a strong likelihood that members of a household were not present as a result of attending school or going to work. For instance, a five-person household which consists of two adults and three children would most likely have members of the household not present during the "business day" (hours constituting the sampler's "office hours"); hence, there is a strong possibility of having higher average inter-use stagnation times during which the RDT sample is taken. After the regular "office hours," members of the family would return home, thus reducing the mean inter-use stagnation time and the lead values (since lead leaching depends on stagnation time) per liter of drinking water; this would be picked up by the measured lead (COMP) sampler but not the RDT sampler. However, the RDT sampler does pick up several inter-use stagnation times, which is very useful. Without the restriction of sampling during office hours only, the RDT method could be closer to the COMP sample.

Another way in which "representativeness" was assessed was by comparing the ratio of lead values from a given sampling protocol and the COMP sample; this value should ideally be equal to one or constant over a wide concentration range to be "representative" (EU Report, p. 39). A prediction range for the ratio was also calculated and test areas were divided into 11 regions (lettered A to K in the

Report). The EU Report concluded for this assessment type that RDT, 30MS with the first liter, 30MS with the second liter, and the average of the two 30 min samples (MSA) perform the best, while the FF generally underestimates the amount of lead in the sample (i.e. the ratio is strictly less than 1). However, the ratio COMP as well as the prediction range for the ratio varied greatly between test areas A to K. In some areas (see p. 42 of EU Report), the FF performed better than the other protocols while it did not perform well in other areas. For instance, in area C (see EU Report, Fig. 19, p. 42) FF is shown to perform better in terms of prediction range for the ratio than 30MS, first and second liters, and their average (30MSA) while in area G it is shown to perform worse than the other protocols and in area A it is performing just as well as the other protocols in terms of size of the prediction ranges. For a given test area when prediction ranges are large (small), prediction ranges tend to be large (small) for all sampling protocols. Only test areas G and H seem highly variable in that all protocols seem to have a greater variation in average ratios compared to lead samples from the measuring device (COMP) although all protocols have a large prediction range. FF appears to be the most accurate sampling protocol among the most highly variable test areas G and H. All other test areas (other than G and H), have relatively the same prediction ranges for the ratios. In other words, within a test area, the accuracy of a sampling protocol is relatively the same for each protocol although we acknowledge that FF does under predict lead in more instances than other protocols when compared to lead showed by the measuring device; sampling protocols are likely to under or over predict lead values simultaneously for a given test area.

The ability of the protocols to detect problem properties was assessed by analyzing the percentage of "positives," "false positives," "false negatives," and "negatives." A test was considered positive if both the protocol sample and the COMP sample produced a value greater than the MCL of 10 μg/L, while a false positive would be considered a case where the protocol indicated a value higher than the MCL when the COMP shows a value less than the MCL. A false negative would mean that the protocol is indicating a value less than 10 μg/L while the measuring device (COMP) is showing a value greater than 10 μg/L. A negative is considered a case where both the protocol and the COMP show a value less than 10 μg/L. From the Report, the percentage of positives (the ability of the protocol to detect problem areas) is highest for RDT followed by the average of RDT and FF, followed by 30MS, and lastly the FF sample taken alone. False positives were also highest for RDT and least for the average of RDT and FF. Once again the average of RDT and FF performed well (or at least as well as the others) but the Report overlooked this in the final assessment.

One explanation for the failure to identify problem properties was that it was "likely to be caused by characteristics of the plumbing system" (EU Report, p. 47). While this may be partially true, the failure to identify problem properties may be due to characteristics in the sampling protocols themselves. The stagnation time in the protocol in particular is likely to identify more problem properties if it is increased (although this would not be cost-effective; see Fig. 2 on page 15 of the Report for relationship between stagnation time and lead) or if stagnation times were adjusted for the number of persons in the household (which would reflect the average inter-use stagnation time).

The EU Report also states that the “protocol should give a realistic estimate of the problems in the area, in order to be able to use it as an effective decision tool. Furthermore, the result should not unnecessarily worry consumers” (EU Report, p. 46). While it is legitimate not to “unnecessarily worry consumers” (the case of false positives), there is also a need to minimize the possibility that a sampling protocol would indicate a value less than 10 μg/L when the measuring device value shows a value greater than the MCL of 10 μg/L (false negatives). For a property identified as a false negative, the consumer believes that his/her drinking water is safe and no action needs to be taken; although he/she will be ingesting unsafe levels of lead each day. On the other hand, with false positives the consumer will be attempting to improve his/her plumbing system but will probably continue to consume “safer” drinking water in the future since the COMP has a value less than the MCL of 10 μg/L. The EU Report has failed to address and analyze the issue of false negatives adequately and has focused its attention on false positives. Furthermore, for the properties which were identified as false positives (sampling lead value > COMP value when in fact COMP < the sampling lead value), the lead concentration from the COMP sampler was very close to the value of 10 μg/L; on average the COMP estimate for lead for false positives was between 6 to 10 μg/L which is very close to surpassing the MCL value of 10 μg/L (see Fig. 26, EU Report, p. 48). Since 80 percent of the properties that were ‘false positives’ had lead plumbing, they were potentially problem properties to begin with. An examination of Fig. 25, which shows the number of false negatives for each sampling protocol, indicates that the number of households with lead over the 10 μg/L is fairly high and is comparable to the number of false positives. For the 30MS and average of RDT and FF, a rough calculation from graphical inspection only shows that approximately 8 percent of households are false negatives, for RDT approximately 4 percent and for FF approximately 15 percent.

“Cost” was based on hypothetical or assumed wage rates, average travel time between properties, time needed by sampler to perform procedure, analysis costs, and write off cost of sampling device. The average time for RDT and FF is much lower than that of 30MS as well as the measured lead from the device as a sampling procedure. The “practicality” of a sampling protocol was based on “several aspects of the procedure (e.g. is the procedure easily applicable, are skilled samplers needed, does the procedure need specific tools…)” (EU Report, p. 56). “Consumer acceptance” although a “very important factor” (EU Report, p. 57) according to the Report was not well defined. The brief description given for this assessment type is that consumer acceptance was based on consumer’s willingness to cooperate and “if the sampling procedure bothers the consumers too much” (EU Report, p. 57). Both practicality and consumer acceptance had no quantifiable evaluation scheme in the Report. However, qualitative results were given. In terms of practicality, cost-effectiveness, and consumer acceptance, the 30MS scored the lowest (worst) of all tested protocols while the RDT scored the highest (best) followed by the FF sampler. RDT had the least cost followed by FF, followed by the 30MS. Lastly, the measured lead from the attached device was three times as expensive as RDT sampling. The measured lead from an attached device (COMP) scored the lowest in terms of cost because of the amount of time involved in the procedure, which made it expensive. However, in terms of social interest the most expensive sampling procedure (COMP) provided the most ideal method of determining lead in drinking water. (Recall that COMP involves attaching a lead measuring device at the consumer's drinking water tap.)

Not only is the EU Report flawed in interpreting the statistics, but it also failed to take into account the highest social opportunity cost or highest social loss associated with total lead in drinking water. For that, a procedure that can detect the highest amount of lead should have been chosen. Based on the information provided in the EU Report (see Fig. 2, EU Report, p. 15), a 6 h stagnation sampling protocol should have been used since this is the period of stagnation that is equivalent to the equilibrium lead concentration (i.e. when lead concentration approaches the saturation level). Since this information is included in the EU Report in Fig. 2, we cannot understand why this information was ignored in the EU Report in recommending the appropriate protocol.

As far as reproducibility is concerned, sampling protocols were assessed by analyzing the relative range which is equivalent to the (max – min) /mean. A relative range of zero is ideal (max − min = 0). The 30MS and the FF sampler performed the best under this criterion while RDT performed the worst. The poor performance of RDT (in terms of reproducibility) was due to the fact that "stagnation time is not controlled for the RDT sample, whereas stagnation time is controlled for both the FF and 30MS samples" (EU Report, p. 52). However, stagnation time is not controlled for the COMP sample as well; the COMP sample can be viewed as a group of individual samples with widely varying stagnation times. Therefore, out of all the protocols, RDT can be best compared to the COMP in terms of stagnation times since samples for the RDT protocol captured widely varying stagnation times (as noted above, results from the RDT protocol could be improved if samples drawn were truly random). The EU Report has also failed to show the results of "reproducibility" for the COMP sample and useful information is lost, such as comparing reproducibility for COMP with the other protocols. "Reproducibility" in the EU Report does not indicate anything about being able to reproduce the results of a sampling protocol repeatedly over different times, but rather it is simply a statistic that shows the range of extremes compared to the mean.

We can summarize our critique of the EU Report as follows:

The RDT and FF protocols together capture more elements of consumer behavior than just a single sampling protocol alone. In fact, statistical results from the EU report show that the average of RDT and Full Flushed provided the most representative sample when compared to other protocols. The average of RDT and FF outperformed the EU's recommended 30MS sampling and RDT. In terms of practicality, cost-effectiveness and consumer acceptance, RDTand FF protocols were evaluated as the two best protocols while the 30MS was judged to have been the worst. All these factors point toward two methods (RDT and FF) which are cost-effective and practical and when analyzed jointly can produce in most cases better results than any single sampling protocol can. The EU report shows that the 30MS protocol underestimates COMP by 20 percent; RDT overestimates by 27 percent, and FF underestimates COMP by 43 percent. It should also be noted that the COMP sample is not a reliable measure of lead as it is the average intake of lead per week but not the maximum possible intake. If the criterion is to avoid the highest social cost of lead, then the COMP sample is also inadequate. From the health point of view, what matters most is the maximum exposure and not the average exposure.

Finally, it should be noted that Danish legislation requires the use of 12 h stagnation. Germany uses 4 h stagnation because that time protects 95 percent of their consumers. Four hours stagnation covers about 80 percent of the maximum saturation concentration of the stagnation curve, while a 30MS covers only 30–40 percent. RDT and the 30MS underestimate the real exposure to lead by 44 and 56 percent, respectively (Hoekstra et al. 2004). This later research, which is also a EU research publication, clearly shows that the 30MS sampling method is completely inappropriate and simply wrong.

10.5 The EPA Sampling Protocol

The guidelines for monitoring requirements for lead in the USA can be found under the US Code of Federal Regulations, Title 40—Protection of the Environment, Chapter 1 Sub chapter D (Water programs) sub part I. Under this EPA guideline, for a residential property, a 1 L sample should be collected at the plumbing system in either the kitchen or sink tap after a *stagnation time of at least 6 h* (nonresidential buildings are required to obtain samples from a tap that is normally used for water consumption). Lead service line samples are collected either (1) at the tap after flushing the volume of water between the tap and lead service line, (2) by accessing the lead service line directly, or (3) by collecting a sample after allowing water to run until a significant change in temperature is felt. The calculation for the volume of water in (1) is based on the interior diameter and length of pipe.

The number of monitoring sites is the same as that which is presented in Table 10.2 for the Canadian Federal guidelines (Canadian Federal guidelines have adopted many of the EPA measures). The EPA also distinguishes between the sizes of water distribution systems. A medium system serves between 3,300 and 50,000 (inclusive) people while small and large systems can be described as those that serve fewer than 3,300 and more than 50,000 people respectively. Table 10.4 shows the frequency with which each system should be monitored. Each sized system is required to monitor fully for two consecutive 6-month periods unless (1) no more than 10 percent of samples are above 15 μg/L for 2 consecutive periods after which they may reduce their monitoring load (e.g. reducing number of samples) or (2) they meet their MCL criteria after initially failing to meet the MCL level, implementing new corrosion control methods as described in the Federal guidelines and retesting problem properties.

Table 10.4 USEPA frequency of monitoring for lead by population size (USEPA 2010)

System size (no. of people served)	First 6-month monitoring period begins on
>50,000	January 1, 1992
3,301–50,000	July 1, 1992
≤3,300	July 1, 1993

In the United States, the difficulties associated with accurately predicting lead levels at short time ranges prompted the EPA in 1992 to put into the regulation a minimum stagnation time of 6 h in sampling protocols for regulatory purposes. The 6 h stagnation time was based on "…a 'worst case scenario' for lead and copper exposure e.g. in the morning after an overnight stand period" (Lytle and Schock 2000, p. 1).

As noted above, the Canadian Federal level guideline, which is based on a 6 h stagnation, is also based on the EPA guidelines. However, Ontario has adopted a 30 min stagnation protocol based on the recommendation of the Ontario Drinking Water Advisory Committee, which itself relied heavily on the EU Report, although there was considerable evidence that 6 h stagnation time most accurately reflected equilibrium lead concentration levels (see Lytle and Schock 2000, Kuch and Wagner 1983, Schock and Gardels 1983, Lilly and Maas 1990). Figure 10.1 shows the groundbreaking work from Kuch and Wagner (1983), which shows the stagnation profile for lead in drinking water. Even at various alkalinities the equilibrium concentration seems to be around the 6 h mark.

Lilly and Maas (1990) have shown that lead leaching is highly nonlinear and that over 60 percent of lead leaching occurs within the first hour, and that up to

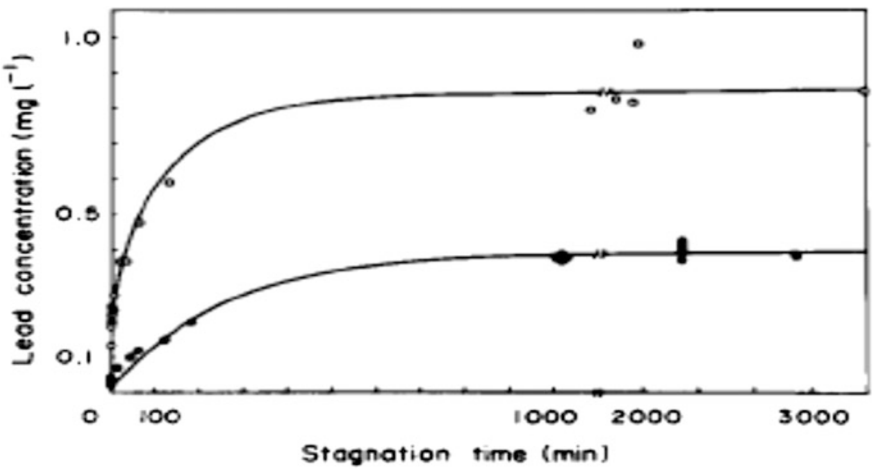

Fig. 10.1 Lead concentration and stagnation time (Kuch and Wagner 1983). *Note Upper line* pipe of ½ inch diameter, pH of 6.8 and alkalinity of 10 mg/L in $CaCO_3$. *Lower line* pipe of 3/8 inch diameter, pH of 7.2, and alkalinity of 213 mg/L in $CaCO_3$

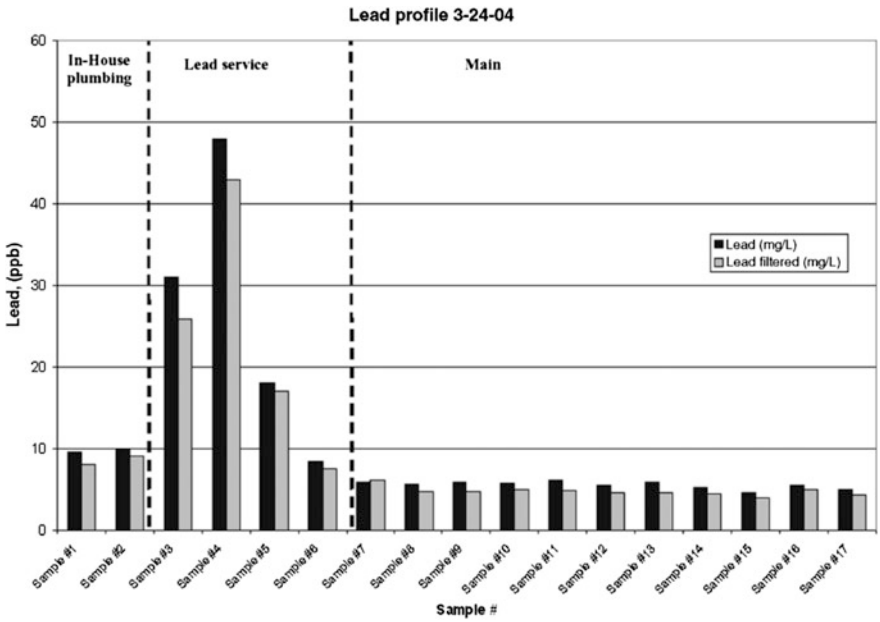

Fig. 10.2 Lead concentration by volume of water drawn in a house in Washington DC (Guidotti et al. 2008)

30 percent of lead leaching can be found in the first 10 min sample. Lytle and Schock (2000) showed that lead leaching accelerates in the first 10 h of stagnation and that up to 70 percent of maximum lead levels can be reached within that time period while leaching can continue to occur even after 90 h of stagnation. The mass transfer model of Kuch and Wagner (1983) indicated equilibrium stagnation time of up to 6 h or more. Lytle and Schock (2000) have advocated obtaining stagnation profiles to predict human exposure and to assess corrosion control treatment. Indeed stagnation profiles can also show peak lead exposure conditions. A profile of the lead concentration by volume of water drawn from a house in Washington DC after overnight stagnation is shown in Fig. 10.2. A similar lead concentration profile was found in a case study for Ottawa (see Fig. 10.3); in the case of the Ottawa samples, both 30 min stagnation and 6 h stagnation profiles are shown.

Figures 10.2 and 10.3 show that lead levels increase sharply when water reaches the tap from the lead service line but decline rapidly once water arrives from the main line section (Guidotti et al. 2008, Campbell and Douglas 2008). Both Figs. 10.2 and 10.3 show that peak lead concentration was drawn at the fourth liter of water while Maximum Allowable Concentration Levels (MAC) were exceeded at the 4th and 5th liters for Ottawa under the 6 h stagnation protocol. Hence, the volume of water drawn in relation to its lead concentration profile can determine lead exposure. Campbell and Douglas (2008) showed that lead in drinking water can be minimized via pH and corrosion control, as well as having a proper

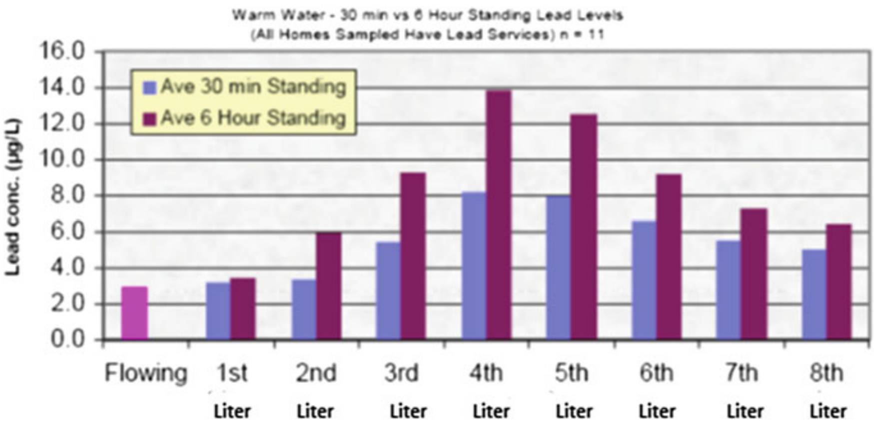

Fig. 10.3 Lead concentration by volume of water drawn for 11 houses in Ottawa (Campbell and Douglas 2008)

understanding of the water chemistry in a distribution area. They concluded that while the current lead sampling protocol (for Ontario) provided a good initial indicator of the typical lead exposure patterns for a given distribution area, additional testing such as analyzing lead concentrations up to the 8th liter can be more useful in detecting locations which are at risk to potentially high lead exposure.

10.6 Conclusion

We can summarize the main findings as follows:

(1) The health effects associated with lead in drinking water are well documented and even at very low-lead levels can cause severe harm to cognitive and intellectual development in young children; (2) Adults are also susceptible to harmful effects from lead, e.g. studies have shown lead can affect kidney function as well as the reproductive system; (3) No threshold effects have been reported for BLLs; (4) The costs associated with BLLs can be quite high. Estimates from Davies (2006) put the cost for lead poisoning for a 1-year cohort of 5-year olds in Washington State at $1478.8 million in 2004 dollars while Landrigan et al. (2002) show a cost of $43.4 billion for a cohort of 5-year olds; (5) Other studies have shown reduced IQ and lower worker productivity; (6) Lead sampling protocols are important in assessing the risk posed from lead in drinking water. Inadequate sampling protocols can result in huge economic losses as stated above; (7) The Ontario sampling protocol used in determining lead in drinking water is heavily based on the EU Report while the new Canadian Federal Guidelines follow the EPA protocol closely; (8) The EU Report recommends either the RDT or 30MS as protocols, which can be used to assess lead levels within a distribution system. However, the Report is faulty in terms of

statistical analysis and the Report itself shows that the combination of RDT and FF methods are better in terms of cost-effectiveness, consumer acceptability, and accuracy (when compared with the COMP sampler) than 30MS or RDT protocol alone; (9) The new Canadian Federal Guidelines give water authorities two options: Option 1 which is a two tier approach where Tier 1 requires a 1 L sample taken after a 6 h stagnation and tier 2 which requires four consecutive 1 L samples taken after 6 h stagnation only if Tier 1 has more than 10 percent of samples above 15 μg/L. Option 2 requires four 1 L samples to be taken after 30 min stagnation; (10) The new Federal Guidelines have accommodated the current Ontario sampling protocol (30 min stagnation) as an acceptable protocol; and (11) The new Federal Guidelines do take into account the usefulness of stagnation profiles in assessing lead corrosion in drinking water but this can only be achieved if Tier 1 of Option 1 fails the protocol eventhough studies show lead levels in excess of 15 μg/L in 4th–6th liters.

We end with the following general conclusion. First, the Ontario lead sampling protocol has followed the European Report and its recommended sampling protocol without realizing the flawed nature of the 30MS protocol even on the basis of the European report. Second, underestimating lead exposure in drinking water can cause huge social losses in lost productivity and lower cognitive development of children and the future labor force. Third, health agencies, scientists, and water policy experts should reexamine the use of the Ontario 30MS protocol and replace it with the 6 h stagnation protocol, as recommended both by the EPA and the Canadian Federal Government Guidelines.

It has been argued that there were logistic difficulties in Ontario in adopting the scientifically correct protocol of sampling after a period of 6 h of stagnation, a protocol that is used by the EPA in the USA. If that is the case, then perhaps the MCL (also called Maximum Allowable Contamination) should be *lowered* in order to capture the risk associated with lead intake. What should that reduced level be? That is the subject of the next chapter.

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Chapter 11
Confronting the Problem of Lead in Drinking Water: What Can and Should Be Done

11.1 Introduction

In the previous chapter, we reviewed the harmful effects of lead in drinking water and highlighted the need to measure lead in drinking water using strict principles of chemistry, and adopt a scientific protocol that is used consistently. The Appendix to this chapter demonstrates statistically that the two sampling protocols under consideration are indeed different, and therefore which sampling method is used does matter in measuring lead accurately.

Adopting a scientifically sound sampling protocol that correctly measures lead in drinking water is the only way of minimizing risk, even low-level current risks that pose long-term health concerns. In this chapter, we attempt to answer two questions: (1) what *can* be done to reduce health risks from lead, and (2) what *should* be done if the 30-min stagnation protocol cannot be changed. To answer the first question, in Sect. 11.1 we turn to a case study of Denmark and outline how it has met the lead challenge. Then in Sect. 11.2 onwards we carry out a statistical simulation and show that if the sampling protocol is the 30-min stagnation used in Ontario, then the obvious thing to do is to *lower* the regulatory maximum contamination level (MCL) of lead, down from 10 μg/L to something less than that. We obtain that lower MCL for Ontario, using Ontario data, in Sect.11.3.4.

11.2 Lead in Denmark

Danish Drinking water is obtained almost exclusively from groundwater. This source accounts for 99 percent of the total water supply. In the majority of water utilities, only aeration and filtration is done before the water is pumped through the delivery system. For the most part, no conditioning is done except for pH adjustment and hardening in some soft water areas. Only a few waterworks use disinfection of

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the water by chlorination, whereas the remaining waters are not treated (Fontenay and Anderson 2008). There are 385 groundwater bodies, distributed within four River Basin Districts.

There are about 2,700 water works that supply the Danish population of 5.5 million. The municipalities which operate 150 water utilities extract 24 times as much water as the 2,550 privately operated water companies (Fontenay and Anderson 2008). Danish authorities have found it unnecessary to cleanse the water with carbon filters or add chloride; only oxidation and cleansing in a sand filter is required before it is drinkable. Approximately 800 million m^3 of water are abstracted annually. Groundwater recharge averages 100 mm per year, varying between 50 and 350 mm.

The standards of wastewater treatment are high. Over 90 percent of the aggregate wastewater is treated in 216 plants, most of which are municipally operated (Danish Ministry of the Environment and GEUS (Geological Survey of Denmark and Greenland 2014). The current Danish groundwater policy is based on protection. The two most common ways are either to protect the resource of natural groundwater or to treat wastewater to the extent that the treated water conforms to the maximum allowable contamination levels (Hasler et al. 2005).

How lead is measured is important and so we repeat the findings of Hoekstra et al. (2004) reported in the last chapter, namely that in measuring the amount of lead in drinking water, the Danish legislation follows the use of 12 h stagnation. Germany uses 4-h stagnation because that time protects 95 percent of their consumers. Four hours stagnation covers about 80 percent of the maximum saturation of the concentration of lead, based on the stagnation curve, while a 30-min stagnation covers only 30–40 percent. Random Day Time sampling and 30-min Stagnation underestimate the real exposure by 44 and 56 percent, respectively.

In Danish groundwater samples taken in the period 1993–2006, lead was found in 406 of 663 (61 percent) abstraction wells and occurred in concentrations over the drinking water standard (5 μg/L, value at the entrance to the property) in 10 samples (2 percent). The average concentration was 0.6 μg/L and the maximal concentration measured was 35 μg/L (GEUS 2007). In 2010, lead occurred in concentrations over the drinking water standard (5 μg/L, value at the entrance to the property) in 4 of 238 samples from Danish groundwater (GEUS 2011).

The Danish Ministry of the Environment has financed metal release projects, where rig testing of commonly used materials was performed at different water works. As a result, taps must be tested for lead and cadmium release by sit-and-soak testing in synthetic, soft water by a Scandinavian standard. This test was introduced in the mid 1970s as a means of controlling whether illegal solders containing lead or cadmium had been used. Nickel-chromium electroplating is accepted without limitation or special requirements (Fontenayand Andersen 2008). Plastic products are examined thoroughly by toxicological tests based on a review of raw materials and production methods with the exception of parts that constitute only a minor part of

Table 11.1 Occurrence of lead pipes in Europe (KIWA 1998)

Country	percent Pb communication pipes	percent Pb supply pipes or internal Pb plumbing
Belgium	19	15–30
Denmark	0	0
France	39	38
Germany	3	9
Greece	<1	0
Ireland	50	51
Italy	2 (?)	5–10
Luxembourg	7	0
Netherlands	6	8
Portugal	?	32
Spain	>3 (?)	?
UK	40	41

the installation. Lead piping has never been used to any large extent, and no lead pipes are in use today either in the main distribution networks or in domestic installations (see Table 11.1).

Where there is very hard water, copper release from copper pipes and zinc release from hot dip galvanised steel pipes are likely to be high after stagnation in some areas of Denmark. For these areas, plastic and stainless steel are the common pipe materials for new installations. Nickel release from taps can be very high in many water types, and the general advice from the Danish Ministry of the Environment (Danish Ministry of the Environment and GEUS 2014) and the waterworks is to discard the first 0.2–0.3 l of water that has been stagnant in the pipes for a long time, e.g. overnight. Studies conducted in Copenhagen, Denmark, found that nickel was leaching from chromium–nickel-plated brass after periods of water stagnation. The Danish Waterworks Association generally recommends flushing until the water is cold after stagnation over a long period, e.g. overnight, to avoid drinking water that may contain some metallic contamination from the pumps, pipes, and home water installations.

Denmark's long campaign to improve water quality has involved many separate initiatives: investing in wastewater and sewage treatment, regulating and reducing the use and thus the discharge of fertilizers from agriculture, banning the use of chemicals that endanger groundwater, and cleaning up deposits of dangerous substances from former times.

It is clear that where there is determination to confront the threat from lead in drinking water, a great deal can be done to enhance public health and safety. The example of Denmark is worth emulating. All that is required is political will, adequate tax and fee revenue, and a sound government administration committed to scientific measurement of lead.

11.3 What the Regulatory Maximum Level of Lead Should Be in Ontario

11.3.1 Overview

In the previous chapter, it was argued that the 30-min Stagnation protocol for sampling lead is inappropriate and wrong. It has been suggested that Ontario adopted this protocol after an Advisory Committee recommended it. But the advisory committee relied largely on the EU Report, which was roundly criticized in Chap. 10. The advisory committee was told of some other constraints as to why the consumer could not be allowed to take a 6-h stagnation sample, which would typically have to be taken by government employees at 6 am.

If the logistic difficulties do not make a 6-h stagnation sample possible, then perhaps the maximum allowable contamination level of 10 µg/L should be lowered. What should that lower level be, if Ontario is constrained to continue to use the 30-min Stagnation sample? We take a statistical simulation approach to answer this question, using the limited available data to carry out some experiments. In other words, we want an estimate of the new lower MCL, as *if the* 6-h stagnation sample had been used. We use data from the City of Ottawa where 6-h stagnation samples have been taken. Unfortunately, in this exercise, the samples were taken *after they had added caustic soda to raise the pH level*, in order to reduce lead from leaching into the drinking water. Hence, the first step is to consider the relationship between lead leaching and pH. We attempt statistically to "reduce" the pH to the Ontario Average, and then use the distribution on lead sampling to simulate what the new reduced "cut off," or MCL should be. The result is instructive and suggests a change in policy, after scientific and chemical validation.

The estimation is carried out in three sections as follows:

(1) Section 11.3.2: The estimation of lead from Ottawa samples that would reflect the average pH values for the rest of Ontario and an analysis of samples before and after pH adjustment.
(2) Section 11.3.3: The simulation of lead for the 'rest of Ontario'[1] 30 min stagnation samples to 6 h stagnation samples and an analysis of samples before and after stagnation time adjustment.
(3) Section 11.3.4: The estimation of the possible lower contamination level.

Then in Sect. 11.4, we present some caveats on the limitations of our simulation exercise.

Recall that our overall aim is to simulate and estimate lead values from samples under the 30-min Stagnation protocol for the 'rest of Ontario' in such a manner that the outcome reflects what it would have been *if* a 6 h stagnation protocol had been

[1] Rest of Ontario refers to samples taken from the Ontario Tap Water Order for 36 Municipalities in 2006 and samples which are not from Ottawa's Customer Lead Pilot testing done in 2006/07. Henceforth, "rest of Ontario" refers to this definition.

used, as the first option given in the Health Canada Guidelines (Health Canada 2009). We utilize data from the 2006 to 2007 Customer Lead Pilot Testing project in Ottawa and assume that the statistical properties for lead from this dataset are representative of the samples for the rest of Ontario. The level of pH affects the dissolution of lead from pipes (i.e. lead being "dissolved") into the drinking water and since the pH levels in Ottawa are much higher than the pH levels for the rest of Ontario, we need to "reduce" the pH levels in the Ottawa data so that it is comparable to the rest of Ontario data. We do this by utilizing data from U-MATE International, which shows the relationship between pH and lead holding all else constant, fitting a functional form to the data and using the functional form to convert Ottawa data to the average pH levels observed in the Ontario data. After the completion of this process, we should have comparable data, in terms of pH, between the rest of Ontario and Ottawa. The next step is to find out what lead values for the Ontario sample would be like if the Ontario samples were taken using the 6-h stagnation protocol. Our first approach is a benchmark model and is a simple percentage change between the 30-min Stagnation and 6-h stagnation protocols, holding pH constant. Since the lead values used in the analysis occur at only two points in time (30 min and 6 h), we assume that regardless of the functional form of the rate of dissolution of lead in water pipes (whether it is linear or not), the percentage change between the two data points is fixed. After the completion of this process, we should obtain estimated lead values from the 30-min Stagnation protocol data from the rest of Ontario "converted" to what lead levels *would have been* had the the 6 h stagnation protocol been used.

11.3.2 The Estimation of Lead

For this subsection, we estimate outcomes for the lead values in the Ottawa data that would reflect the pH values in Ontario. The range of pH for Ottawa data (under the "City of Ottawa Customer Lead Pilot Testing 2006–2007" project) was between 8.57 and 9.46 while the range of pH for Ontario data (under the "Ontario tap Water Order for 36 Municipalities" in 2006) was between 6.37 and 8.4. Therefore, we need the two datasets to be comparable in terms of pH. We utilize data from U-MATE International that shows the relationship between pH levels and lead (see Fig. 11.1) and fit a nonlinear functional form.

Figure 11.1 above shows a similar functional form between lead and alkalinity in Schock's (1989) analysis of temporal variability in domestic plumbing systems (replica graph from Schock (1989) is shown in Fig. 11.2).

Our estimated functional form is that of a quadratic with the estimated equation being: $y = 0.003x^2 - 0.0643x + 0.3622$, with an R-squared value of 0.9247, where y is the lead value and x the pH value.

A fitted cubic functional form for the U-MATE data did not show any significant difference both in terms of slope and R-squared values. We also did not use higher-order polynomials (above order 3) since the interpretation of coefficients in higher

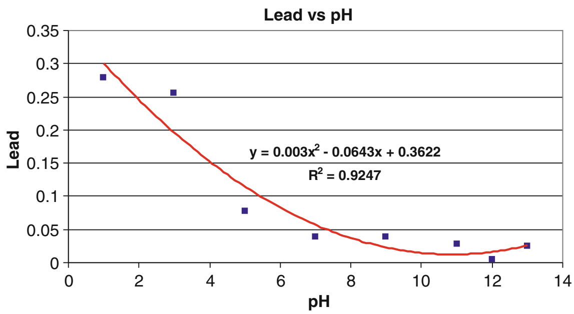

Fig. 11.1 Lead versus pH with fitted functional form (according to the data from U-MATE International)

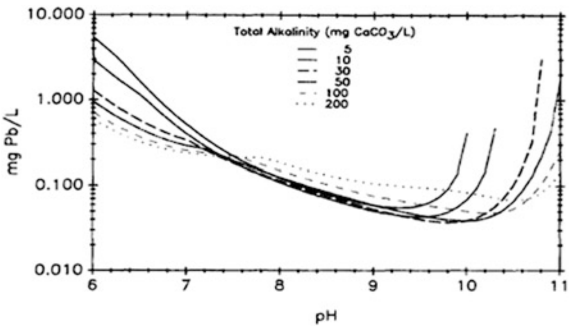

Fig. 11.2 Schock's (1989) analysis of lead versus alkalinity is similar to the functional form observed in U-MATE data (the thermodynamic data used is from Schock and Wagner (1985)

orders is not very meaningful; also we did not want to over fit the dataset that was quite small. Estimated fitted values (not shown) for lead for higher polynomial orders were sometimes negative, and therefore the fitted higher order polynomial functions were not practically useful.

We then took each of the pH values for Ottawa and obtained the fitted 'y-values' or lead values according to the quadratic equation. We did the same for the average pH of the rest of Ontario data. The associated lead value for each pH value of the Ottawa data would then be increased by the percentage difference between y-values for Ottawa and the y-value for Ontario. In this way, we could obtain Ottawa lead values with a pH of 7.4. For instance, from Fig. 11.3, if we take Ottawa pH of 9.57 and use the fitted quadratic functional form, we can estimate the lead value if its pH was 7.4 (the Ontario average).

Figures 11.4, 11.5, 11.6 and 11.7 show the results of the estimation process of Sect. 11.3.2. The Ottawa data are also grouped into Spring 2007, Summer pre water-main rehab 2007, and Summer post water-main rehab 2007. This is for the first liter only with a stagnation time of 6 h.

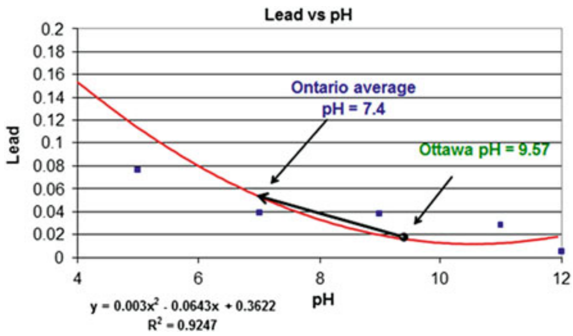

Fig. 11.3 An example of scaling the Ottawa pH values and its associated lead values to the Ontario average

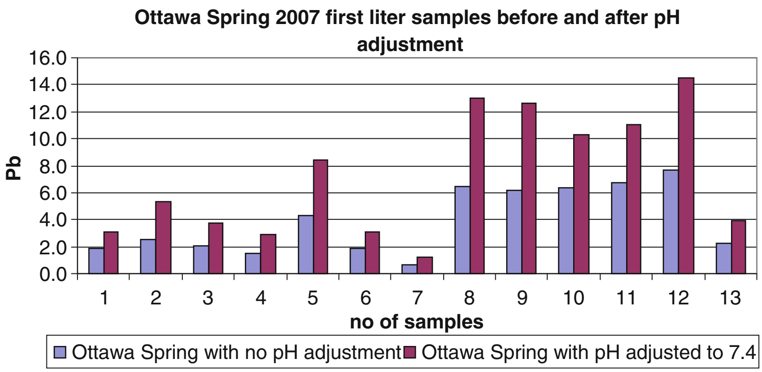

Fig. 11.4 Ottawa spring 2007 first liter samples before and after pH adjustment

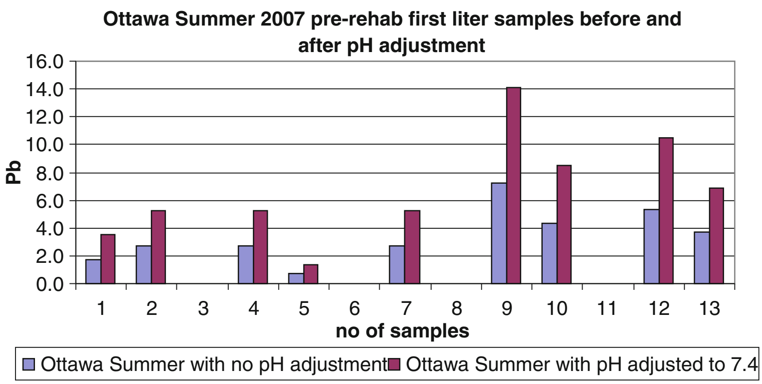

Fig. 11.5 Ottawa summer pre-rehab 2007 first liter samples before and after pH adjustment

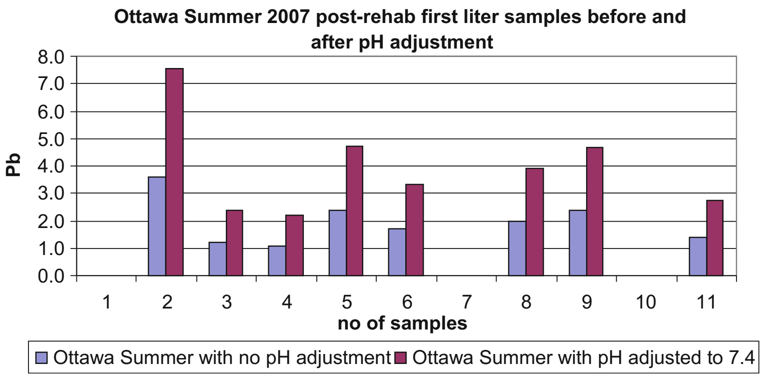

Fig. 11.6 Ottawa summer post rehab 2007 first liter samples before and after pH adjustment

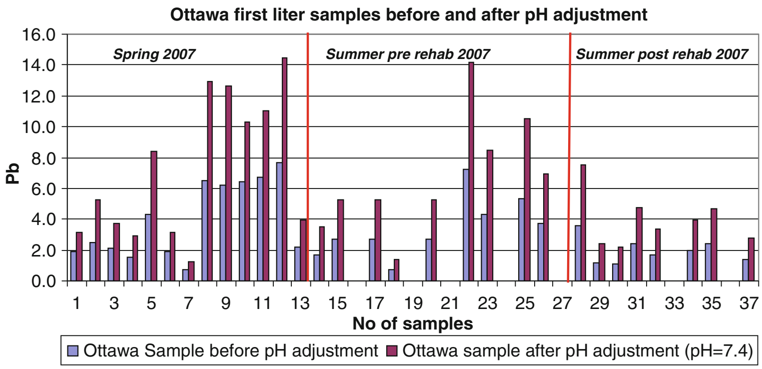

Fig. 11.7 Ottawa first liter samples (all) before and after pH adjustment

Prior to the pH adjustment, Ottawa had no first liter samples over 10 μg/L. After the pH adjustment (i.e. adjusted lead values to pH = 7.4), Ottawa had five samples over 10 μg/L in Spring 2007, two samples in Summer pre-water main rehab 2007 and no samples in Summer post water main rehab 2007.

One anomaly in the Ottawa data is that Spring 2007 first liter samples showed higher lead values than Summer (pre- and post-water main rehab) 2007 samples even before the pH adjustment although temperature should increase the rate of dissolution of lead. Location can be a reason for this as each home plumbing system is unique and will produce different results depending on many internal plumbing factors. Another reason could be due to temperature itself. We would expect higher temperature in the summer (average for Ottawa data is 22.6 C) to be associated with higher lead values in water samples and lower temperature in spring (average for the Ottawa data is 2.2 C) with lower lead values. However, since we are examining the first liter only, that sample would be taken from the internal plumbing within the home (from the tap and most likely piping with the home). During the spring, homes are heated; note the spring average temperature is 2.2 C on average from the Ottawa data and it is highly likely that the home was heated at the time of sampling. So while the external temperature (e.g. 2.2 C) is low, the internal temperature of the home can be quite high. The opposite occurs during summer months when homes are cooled rather than heated. This can explain the high lead levels for the first liter (only) in spring versus the low lead levels in summer.

11.3.3 The Simulation of Lead Samples

In this subsection, we use information from the estimated lead values obtained in Sect. 11.3.2 to simulate samples for a 6-h stagnation protocol for the 'rest of Ontario' data. We use a simple linear approach as our benchmark model. First, we use the same approach as in Sect. 11.3.2 to estimate the lead values for first liter sample for the rest of Ontario (30-min Stagnation) such that all samples have a common pH value of 7.4.[2] After the completion of this process, we would have Ottawa data under 6-h stagnation protocol with a pH of 7.4 and the rest of Ontario data under the 30-min Stagnation protocol also with a pH of 7.4. To obtain 'rest of Ontario' lead values in a manner that would reflect a 6 h stagnation, we take the rate of dissolution of lead as the average percentage change between the '6 h' samples and '30 min' samples. This percentage change (an increase) will be applied to the lead values for the '30 min' samples in order to obtain 'new samples' as if taken under the 6-h Stagnation protocol. We are not specifying the functional form for the rate of dissolution of lead but merely assuming the percentage change in lead between the 6 h samples and 30 min samples is a fixed amount and does not depend on the path of the functional form (linear or nonlinear) from time at 30 min to time at 6 h (see Fig. 11.8 for an example).

Figures 11.9, 11.10 and 11.11 show a summary of the results from the simulation of Ontario first liter samples under the 6-h stagnation protocol. The percentage change factor was obtained in three ways: (1) Using all of Ottawa data—Spring 2007, Summer 2007 pre water main rehab and Summer 2007 post water main rehab, (2) Ottawa spring 2007 data only and (3) Ottawa summer data only (pre- and post-water main rehab).

The number of samples above 10 μg/L for the rest of Ontario under the 30-min stagnation protocol was 52. Under the 6-h stagnation protocol, the number of samples above 10 μg/L is 231 using all Ottawa data. If we base the percentage increase in lead values from 30 min stagnation to 6 h stagnation using only Ottawa summer data, the number of samples for the rest of Ontario above 10 μg/L is 214 compared to that of 253 when basing the percentage increase using Ottawa spring

[2] Although the Ontario average pH is 7.4, individual samples would be higher or lower than the average. Hence it is beneficial to have a common pH value for a consistent analysis.

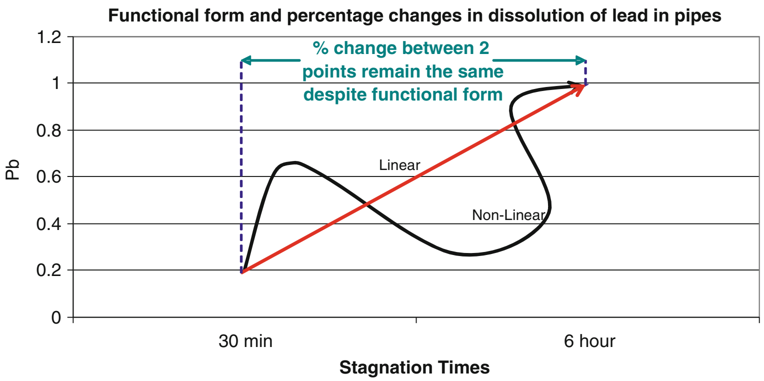

Fig. 11.8 Functional form of lead dissolution and percentage change between two points

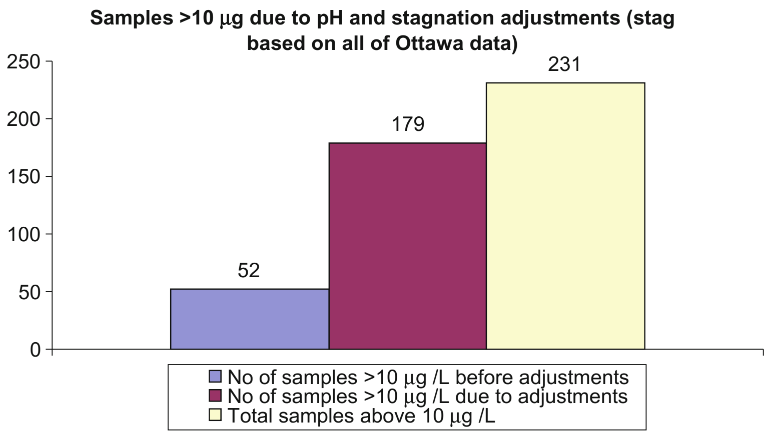

Fig. 11.9 Number of samples greater than 10 μg/L after pH and stagnation adjustment for Ontario

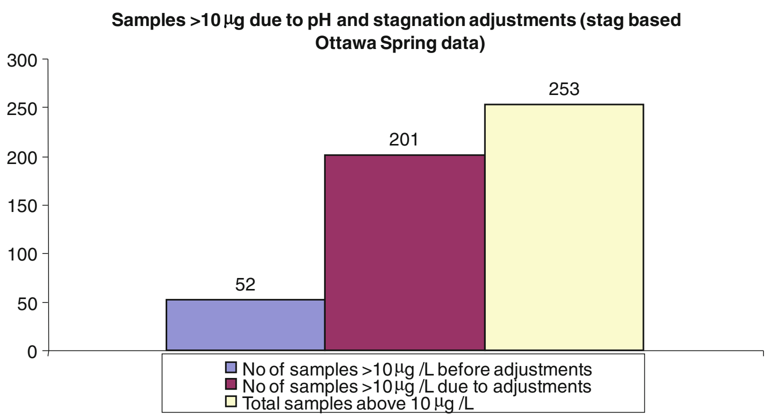

Fig. 11.10 Number of samples greater than 10 μg/L after pH and stagnation adjustment for Ontario

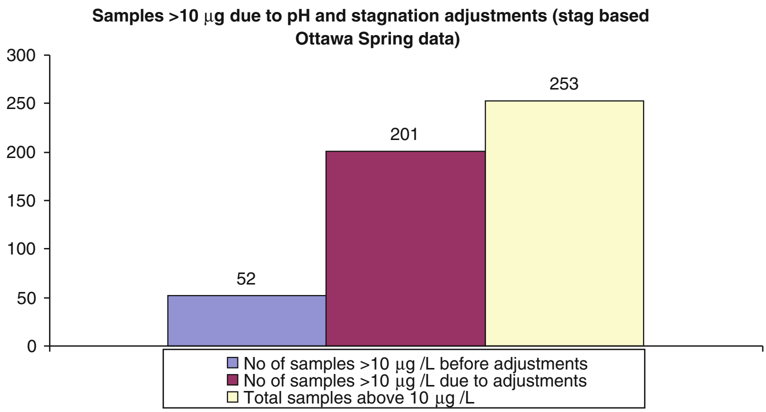

Fig. 11.11 Number of samples greater than 10 µg/L after pH and stagnation adjustment for Ontario

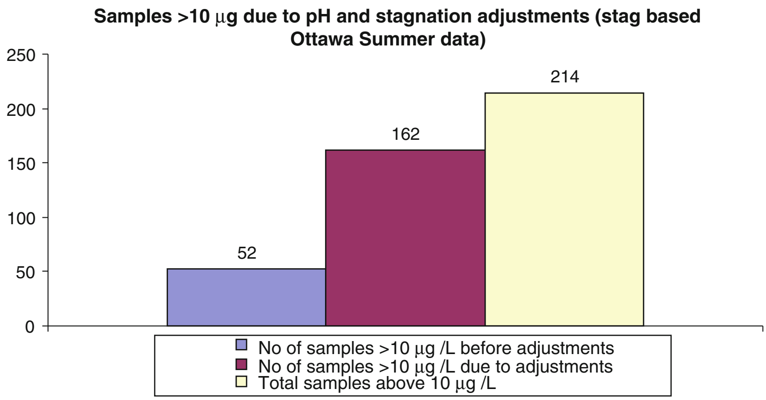

Fig. 11.12 Number of samples greater than 10 µg/L after pH and stagnation adjustment for Ontario. *Note* pH in Ottawa ranges from 8.6 to 9.5; pH for the rest of Ontario ranges from 6.4 to 8.4

data only.[3] We have more confidence in the estimated lead values when the percentage increase in lead between the 30 min and 6 h stagnation is based on Ottawa summer data only since the temperature values for Ottawa summer are closer to that observed in the Ontario data.

Figure 11.12 shows all samples from the Ottawa data under 30-min stagnation protocol versus 6-h stagnation protocol.

[3] See Sect. 2.1 for explanation on temperature differences and the first liter sampling.

11.3.4 Simulating the Lower MCL for Lead for Ontario

We now proceed to estimate the probable MCL given 6-h stagnation simulation results:

11.3.4.1 Hypothetical Experiment 1

Assume 231 samples of a total of 1,352 were indeed exceeding 10 μg/L: this is approximately 17.5 percent of data. Assume further that this proportion of samples above an MCL is representative of the population.

Then for the data for Ontario under the 30-min Stagnation protocol, what is the MCL level given these assumptions?

Order Ontario data under 30-min Stagnation for lead from lowest to highest; remove top 17.5 percent; assume that top 17.5 percent of samples are above MCL as in 6 h stagnation. This results in a cut-off point of 2.8 μg/L. Samples above this value can be seen as above the hypothetical MCL. This is our first estimate of what the Maximum Allowable Contamination or MCL by Lead should be, IF we must use the 30-min Stagnation protocol.

11.3.4.2 Hypothetical Experiment 2

Next, use statistics to obtain minimum and maximum lead below 10 μg/L under 30-min Stagnation protocol that would become greater than 10 μg/L after 6-h stagnation as shown in Table 11.2.

Most restrictive (most risk averse) MCL estimate: Sample with minimum value for lead under 30 min stagnation protocol but which would be above 10 μg/L after a stagnation time of 6 h is 2.7 μg/L.

Moderately risk averse MCL estimate: (1) Half of samples had lead values below 5 μg/L before adjustments were made, and (2) Average lead value for samples were 5.4 μg/L before stagnation adjustment. This is a workable average.

Very "status quo" oriented, and possibly risky MCL estimate: Samples with maximum lead value of 9.9 μg/L would be the new "cut off," given the samples that

Table 11.2 Statistics for samples below 10 μg/L before adjustments but above 10 μg/L after adjustments

Min	2.7
Max	9.9
Median	5.0
Average	5.4
No. of samples	179

were found to exceed 10 μg/L after stagnation adjustment: this is not a very surprising result.

The conclusion of this statistical simulation is that if, for logistic reasons, Ontario must continue to rely on the 30-min Stagnation protocol (say, because all sampling must be done by authorized staff who cannot be compelled to take a 6-h Stagnation sample at 6 am), then the sensible action for the Government would be to *lower the Maximum Allowable Contamination or MCL for lead to the 5.0–5.4 μg/L range.*

11.4 Some Caveats and Limitations

We acknowledge that there are more variables that influence the amount of lead present in tap water. However, for our purposes pH and time are two of the most relevant. Our focus on pH as opposed to temperature stems from the fact that Ottawa has implemented measures (e.g. adding $CaCO_3$, and sodium hydroxide, and/or CO_2) to increase pH levels in water as a result of tests that had shown lead levels *greater* than the MCL of 10 μg/L. Although temperature is an important variable that can influence lead levels in pipes, our simulation exercise makes it difficult to incorporate another variable. We also understand that each sample is taken from a unique plumbing system within the home and that the lead levels can vary for a great number of reasons. However, we are assuming that the data is representative of the entire population and that the estimation of lead for Ontario with a 6-h stagnation protocol is a result of a thought experiment and should be treated as such. In Chap. 10, we presented detailed criticisms of Van den Hoven et al. (1999), a European Union report that supports the 30 min stagnation on the grounds of practicality, costs, reproducibility, representativeness and consumer acceptability. For the social point of view, we need to find the most representative lead exposure, which in Van den Hoven Report is the *composite proportional sample (COMP).* The maximum exposure, even according to Van den Hoven, is 6 h of stagnation. The average of the random daytime sample and the fully flushed sample shows a better correlation with COMP that the 30-min stagnation. Both underestimate lead with reference to COMP but the average of the random daytime sample and the fully flushed sample underestimates to a lesser degree than the 30-min stagnation sample. But as reported above, another EU report (Hoekstra et al. 2004) shows that the 30-min Stagnation underestimates lead in drinking water by as much as 54 percent. Therefore, our results based on a statistical simulation for the *lowering* of the recommended Maximum Contamination Level of lead to something close to what Denmark allows at the entrance of the property seems reasonable, and that level is 5 μg/L.

Appendix

New Federal Guideline Option 2 (30-min stagnation, four 1 l samples) compared to new Federal Guideline Option 1 (6 h stagnation, one 1L sample—Tier 1, four 1 L samples—Tier 2)

The objective of this Appendix is to check if the two main lead sampling protocols are the same or whether they are different. If they were the same, then it would not make any difference whether the 6-h stagnation protocol was used or whether to 30-min stagnation protocol was used. To do this statistical test, we consider the entire distribution of each and test to see if the proportion of samples greater than 10 μg/L is the same in the two distributions. To carry out this test, we use a two-tailed test.

(1) The proportion of samples >10 μg/L for Option 2 (1 to 4 L, 30 min stagnation) versus proportion of samples >10 μg/L for Option 1 (6 h stagnation, one 1 L sample, Tier 1, four 1 L samples, Tier 2)

   (a) Consider each sample as a separate sample in Table 11.3.

Next, see Table 11.4.

Since the null hypothesis is rejected (with a p-value of zero), we can conclude that the proportion of lead above 10 μg/L for samples (1st to 4th liter) from Ottawa 6 h stagnation (Option1—Tier 2) is significantly different from the proportion of

Table 11.3 Separate samples

	Ottawa first liter samples 6 h stagnation (Option1—Tier 1)	Ottawa 1 to 4 L samples 6 h stagnation (Option1—Tier 2)	Ottawa 1 to 4 L samples 30 min stagnation (Option 2)
Total number of samples	30	120	316
Number of samples >10 μg/L	0	22	14
Proportion of samples where lead >10 μg/L	0 percent	18.3 percent	4.43 percent

Table 11.4 Testing the hypothesis that the proportion of samples above 10 μg/L is different (Null hypothesis: the population proportions are equal)

Proportion of samples above 10 μg/L	Two-tailed p-value
Ottawa 1 to 4 L samples 30 min stagnation (option 2) versus Ottawa 1 to 4 L samples 6 h stagnation (option 1—Tier 2)	0.0000
Ottawa 1 to 4 L samples 30 min stagnation (option 2) versus Ottawa first liter samples 6 h stagnation (option 1—Tier 1)	NA[a]

[a] Due to no first liter samples above 10 μg/L

Table 11.5 Four consecutive liters under options 1 and 2

	Ottawa first liter samples 6 h stagnation (option 1, Tier 1)	Ottawa 1 to 4 L samples 6 h stagnation (option 1, Tier 2)	Ottawa 1 to 4 L samples 30 min stagnation (option 2)
Total number of samples	30	30	79
Total number of samples >10 μg/L	0	22	14
Proportion of samples where any liter contains lead > 10 μg/L	0 percent	73.3 percent	17.72 percent
Number of samples where avg lead >10 μg/L	0	3	3
Proportion of samples where avg lead >10 μg/L	0 percent	10 percent	3.80 percent

Note 17.7 percent of locations have lead over 10 μg/L in at least one of the 4 l of drinking water under the 30-min stagnation protocol while 73 percent of locations have lead over 10 μg/L in at least one of the 4 l of drinking water under the 6 h stagnation protocol. 3.8 percent of locations had average lead for first four samples above 10 μg/L under the 30 min stagnation protocol while 10 percent of locations had average lead for the first four liters above 10 μg/L under the 6 h stagnation protocol (option 1, Tier 2)

lead above 10 μg/L for samples from Ottawa (1st to 4th liter) 30 min stagnation (Option 2) assuming each sample is independent of each other. Therefore, we can say that the sampling protocol *does* matter.

(b) Consider the four consecutive liters under Options 1 and 2 as one sample in Table 11.5.

Now consider the test in Table 11.6.

Since the null hypothesis is rejected (with a p-value of zero), we can say that the sampling protocol *does* matter. We can conclude that the proportion of lead above 10 μg/L for samples (1st to 4th liter) from Ottawa 6 h stagnation (Option 1) is significantly different from the proportion of lead above 10 μg/L for samples from Ottawa (1st to 4th liter) 30-min stagnation (Option 2) assuming the first to the fourth samples are considered a collective sample.

(2) Comparison of the average for samples (1st to 4th liter) from Ottawa 6-h stagnation (Option1) versus the average lead for Ottawa (1st to 4th liter) 30-min stagnation (Option 2)

(a) Consider a comparison of Option 2 Versus Option 1, Tier 2 in Table 11.7.
(b) Now consider a comparison of Option 2 and Option 1, Tier 1 in Table 11.8.

Table 11.6 Testing the hypothesis that the proportion of samples above 10 μg/L is different (Null hypothesis: the population proportions are equal)

Proportion of samples above 10 μg/L	Two-tailed p-value
Ottawa 1 to 4 L samples 30 min stagnation (option 2) versus Ottawa 1 to 4 L samples 6 h stagnation (option1—Tier2)	0.0000
Ottawa 1 to 4 L samples 30 min stagnation (option 2) versus Ottawa first liter samples 6 h stagnation (option1—Tier 1)	NA[a]

[a] Due to no first liter samples above 10 μg/L

Table 11.7 Comparison of option 2 versus option 1, Tier 2

	Ottawa 1 to 4 L samples 30 min stagnation (option 2)	Ottawa 1 to 4 L samples 6 h stagnation (option 1—Tier 2)
Total number of samples	316	120
No. of samples >10 μg/L	14	22
Min	0	0.3
Max	16.9	31.0
Average lead (μg/L)	3.5	6.4

Note Average lead under Option 1, Tier 2 is almost 2 times greater than that of Option 2 under the new Federal guidelines. An increase of 8 samples above 10 μg/L is observed when 6-h stagnation protocol is used instead of 30-min stagnation protocol

Table 11.8 Comparison of option 2 versus option 1, Tier 1

	Ottawa 1 to 4 L samples 30-min stagnation (option 2)	Ottawa first liter 6-h stagnation (option 1—Tier 1)
Total number of samples	316	30
No. of samples >10 μg/L	14	0
Min	0	0.7
Max	16.9	7.7
Average lead (μg/L)	3.5	3.2

Note Option 1, Tier 1 samples had no samples with lead over 10 μg/L; 14 were found for Option 2. Recall that 22 samples were above 10 μg/L for the first 4 l under 6-h stagnation. Since Option 1, Tier 1 samples had no samples with lead over 10 μg/L, the 22 samples over 10 μg/L came from samples 2 L to 4 L under the 6-h stagnation protocol (Tier 2). That is, even though we may not find problem properties or houses associated with Option 1, Tier 1 there can be problem properties or houses when Tier 2 is applied. However, Tier 2 is not applied unless more than 10 percent of sites have lead over 15 μg/L under Tier 1. Therefore, there can be many properties or houses that have high lead concentrations in later liters but we may never find out that this is the case since Option 1 Tier1 may never detect problems via the first liter only. Option 1, Tier 1 underestimates the extent of the lead corrosion problem

Table 11.9 Testing for differences between averages

Average lead values for:	Two-tailed p-value assuming no common population standard deviation
Ottawa 1 to 4 L samples 30 min stagnation (option 2) versus Ottawa 1 to 4 L samples 6 h stagnation (option 1—Tier 2)	0.0000
Ottawa 1 to 4 L samples 30 min stagnation (option 2) versus Ottawa first liter 6 h stagnation (option1—Tier 1)	0.5567

Finally, we test for the statistical difference between the averages (testing whether or not average lead values for Ottawa 1 to 4 L samples 30-min stagnation (Option 2) are statistically significantly different from those of Ottawa 1 to 4 L samples 6-h stagnation (Option1—Tiers 1 & 2), Null hypothesis: Difference of means = 0) in Table 11.9.

We can conclude that the mean lead value for Ottawa's first liter 6-h stagnation one-liter sample is not statistically different from the mean lead value for first liter 30-min stagnation sample since the null hypothesis is accepted (with a p-value of 0.5567). That is, the sampling method does not make any difference for the average lead under Option 2 and Tier 1 (or all the samples come from the same population). However, since the null hypothesis is rejected (with a p-value of zero), the mean lead value for Ottawa 1 to 4 L samples (6-h stagnation) is statistically different from the mean lead value for Ottawa 1 to 4 L samples (30-min stagnation), indicating the sampling protocol *does* matter.

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Part IV
A European Case Study

In this case study, we report on how water is managed in Germany, but we also draw on the experiences of Denmark and the Netherlands. We attempt to answer the following questions:

- Why have the European countries switched mainly from surface water to groundwater sources for their drinking water?
- How is water production organized in Europe?
- Why do Europeans emphasize "ecosystem" health rather than just "human health," as in North America?
- What is the social objective behind the "high" price of drinking water in Germany?
- How is water and wastewater priced in Germany?
- To what degree is wastewater treated in Germany?
- What is the state of the water infrastructure in Germany?
- What is the approach to "micro-pollutants" (i.e. pesticides, pharmaceuticals, and personal care products) in drinking water in Germany and the Netherlands?
- What is the quality of treated drinking water in Germany?

Chapter 12
Drinking Water in Germany: A Case Study of High Quality Drinking Water

12.1 Introduction

In Germany, two important federal laws, the *Federal Water Act* (1957) and *the Waste Water Charges Act* (1976), constitute essential elements of water resources and wastewater management. In 2002, when the Seventh Amendment to the *Federal Water Act* came into force, "Germany completed the transposition of the *European Water Framework Directive* (2000) into federal framework legislation, thereby creating the basis for achieving the EU-wide objective of good status for all bodies of water" (German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU) 2014). According to the federal *Waste Water Charges Act* (last amended in 2005) and supplementary provisions of the Germany federal Länder (i.e. the individual states within the Federation), "charges must be paid for waste water discharged into water bodies", and water and wastewater are legally bound to comply with the principles of polluter pays and full cost recovery (BMU 2014) and Bauby (2011). The goal of the *Act* was to reduce the quantity of discharged water to a minimum. In addition, the majority of the Länder (or states) also levy charges for groundwater abstraction, and some also for abstraction from surface water bodies. The management of water resources covers whole river basin districts, including the Danube, Rhine, Maas, Ems, Weser, Elbe, Eider, Oder, Schlei/Trave, and Warnow/Peene (BMU 2014).

Currently, the German water industry has achieved high performance, including long-term safety of supply and disposal, high drinking water quality, high wastewater disposal standards, and high customer satisfaction (Association of Drinking Water from Reservoirs (ATT) et al. 2011). Thus the purpose of this chapter is to describe Germany's approaches in water management. The second and third sections provide a profile of drinking water supply and water consumption in Germany, respectively. Since wastewater treatment technology in Germany ranks the highest in the world, the purpose of Sect. 12.4 is to demonstrate the current status of German wastewater treatment and provide an overview of the

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Springer Water, DOI 10.1007/978-3-319-11032-5_12

development of wastewater treatment technology in Germany. In addition, the safety of water supply and disposal relies on continual investment in maintenance and renewal of the infrastructure. In Germany, all investment costs are included in prices and charges.

The presence of pharmaceuticals and personal care products (PPCPs, or micropollutants) originating in wastewater that typically end up in surface waters that themselves are a source of drinking water has become an issue of global concern. However, the level of wastewater treatment in parts of Europe has been exemplary, and accordingly Sect. 12.5 covers how the problem of micropollutants is handled in the Netherlands, the USA, and Germany.

In Sect. 12.6, we explore the cost structure of water supply and wastewater discharge, and in Sect. 12.7 we discuss water pricing in Germany. Section 12.8 shows the important effects of benchmarking in the water industry, leading to a significant improvement of water supply security, water quality, and sustainability in Germany. Finally, we compare the German Maximum Concentration Levels (MCLs) for drinking water with those of Ontario in Sect. 12.9.

12.2 Drinking Water Supply

12.2.1 Introduction to Drinking Water Utilities in Germany

In Germany, around 99 percent of the population is served by public water supply from over 6,200 water utilities, while about 700,000 citizens are supplied with water from private sources such as those assigned to private houses or villages (ATT et al. 2011). Of the total of 6,200 water utilities, approximately 3,500 are public utilities and the remainder are private businesses (see Fig. 12.1). Most utilities, in particular the smallest ones, are owned by municipalities as a single service or as part of a multi-service municipality, where the small utilities provide only 25 percent of the water by volume (Althoff 2007). In Germany both public companies and private companies are involved in public water supply. These companies can be subdivided into mixed public–private companies, "public-law" companies, water and soil associations, special-purpose associations, municipal companies, as well as other "private-law" companies (Althoff 2007). According to data from the German Association of Energy and Water Industries, in 2008, the larger volume of drinking water was provided by private companies, which made up 64 percent of the total water volume, while public companies provided 36 percent of the water volume (see Fig. 12.1).

In Germany, water supply and wastewater management are the responsibilities of municipalities or other public corporations (Bauby 2011). The Drinking Water Ordinance of the Federal Ministry of Health governs the quality of drinking water, and it is enforced at Länder level (BMU 2014). Under the Drinking Water Ordinance, drinking water is monitored regularly at short intervals and complies with strict quality requirements. In 2010, even Nitrate concentration, the most problematic parameter, now complies with the limit value of 50 mg/L (BMU 2014).

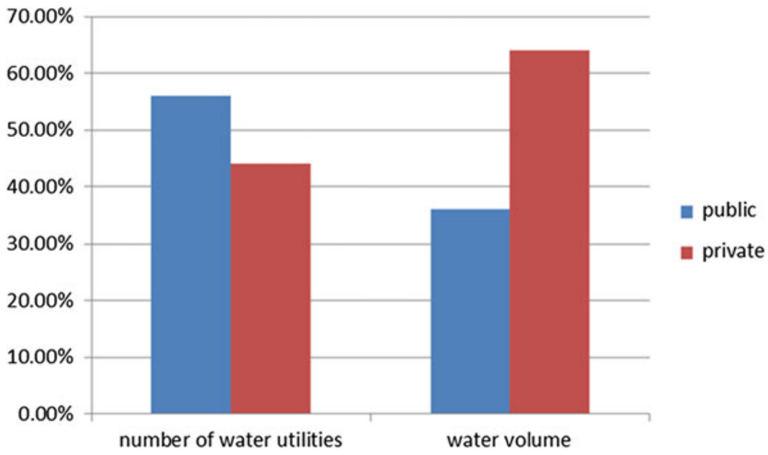

Fig. 12.1 Companies for public water supply in 2008 (Reproduced from ATT et al. 2011)

As a result, according to the Federal Ministry for the Environment, the quality of drinking water in Germany is very good. More than 91 percent of customers are extremely satisfied or satisfied with the water quality (ATT et al. 2011).

12.2.2 Groundwater and Surface Water Bodies in Germany

Groundwater reserves are the most important source of drinking water. Roughly 74 percent of drinking water is drawn from ground and spring water, and the remainder is drawn from surface water sources, such as lakes and rivers Althoff (2007). Article 7 of the *EC Water Framework Directive* requires that "member states shall ensure the necessary protection for the bodies of water identified with the aim of avoiding deterioration in their quality in order to reduce the level of purification treatment required in the production of drinking water." Moreover, the objective of Article 8 of the *EC Water Framework Directive* is to achieve a "good ecological and chemical condition" of surface water and a "good quantitative and chemical condition" of groundwater by 2015.[1] In Germany, the Länder are responsible for implementing water legislation and water protection measures for all water including groundwater. All source waters are monitored through a comprehensive monitoring network to test for contamination under the Federal Water Act.

By 2010, 63 percent of the groundwater bodies in Germany had achieved a rating of "good chemical status" (BMU 2014). Of the total 1,000 groundwater bodies, only 4 percent have not achieved a "good quantitative status," i.e. 4 percent

[1] The regulations set out in the European Water Framework Directive have been incorporated into German law with the Federal Water Act (BMU 2014).

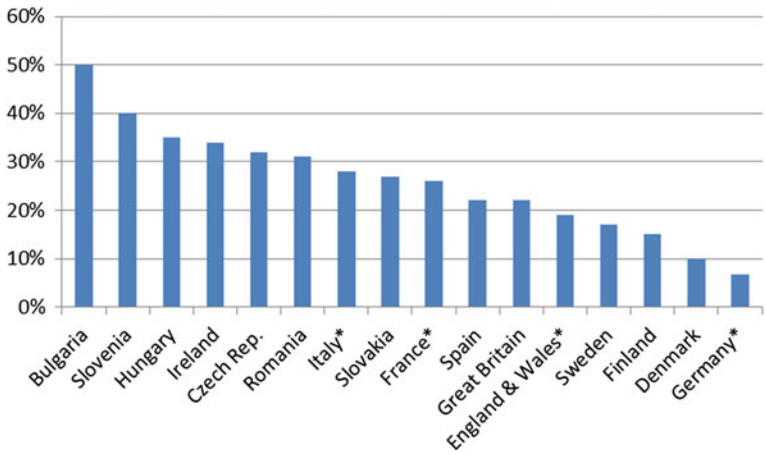

Fig. 12.2 Water losses in the public drinking water networks in EU countries (VEWA-Studie 2006, as cited in Althoff 2007) *Note* * Extractions for operational purposes and fire control are rated as losses

of the aquifers did not have enough water. The status of surface water is such that 88 percent of water bodies achieved a "good" chemical status, while only 10 percent of all surface water bodies had obtained at least a "good" ecological status (BMU 2014). Over the past 10 years, heavy metal pollution as well as pollution from organic pollutants (i.e. benzene, PCB, chlorine pesticides or organic compounds) of surface water bodies has decreased significantly; this has led to an increase in oxygen concentration that is vital for the survival of fish and other aquatic animals. However, there is still room for improvement of surface water quality in Germany.

12.2.3 Security of Supply

Compared to other European countries, Germany has high technical standards of treatment and distribution as well as a well-maintained distribution network of pipes. The water losses caused by burst pipes and leakage have reduced considerably from 600 to 495 millions of cubic meters during the period 1990–2004. As a result, German citizens have not experienced a long-term interruption of water supply. Compared to other European countries, water losses[2] in Germany are 6.8 percent, which is the lowest rate of loss in Europe, followed by Denmark with 9 percent. The low water losses in Germany are due to investments into

[2] It should be noted that water loss is a most important indicator of network quality and security of supply.

maintenance and renewal of infrastructure. In Italy, France, as well as England and Wales, water losses amount to 28, 26, and 19 percent, respectively (see Fig. 12.2).

On average, the rates of damages to water supply lines are less than10 damages per 100 km per year. Moreover, the total investment in drinking water supply amounts to more than 2 billion euros per year. The investments are financed through higher prices and charges that also cover facility maintenance. Due to a stable population, there appears to be no need for an extension of the water network. Furthermore, Germany has the highest average investments in the drinking water sector. In the period from 1995 to 2003, Germany invested 0.54 euros per cubic meter, while England and Wales invested 0.53 euros per cubic meter, France 0.33 euros per cubic meter and Italy 0.15 euros per cubic meter in the same period (VEWA-Studie 2006, as cited in Althoff (2007).

12.3 Water Consumption in Germany

Germany is a water-rich country. In 2007, the total renewable water reserve amounted to 188 billion cubic meters. Only 17 percent of total annual water reserve (or 31.8 cubic meters) was actually used, of which 10 percent (or 15.6 cubic meters) went to thermal power plants for public supply, 3.8 percent (or 7.1 cubic meters) to the mining and manufacturing industries for industrial purposes, 2.7 percent (or 5.1 cubic meters) for water utilities to provide drinking water to households and small business enterprises (see Fig. 12.3). Moreover, in Germany water abstraction for agriculture plays a minor role, with less than 1 percent of the total available water resources, thanks to climatic and geographic conditions, while the other EU countries have a much higher water consumption for agriculture.

As shown in Fig. 12.4, over the past two decades, drinking water consumption has fallen by 17 percent to 122 liters per capita per day in 2009. The drop was primarily due to (a) conservation and water pricing policies that include incentives for efficient use of water resources, (b) the linking of the price to the amount of water

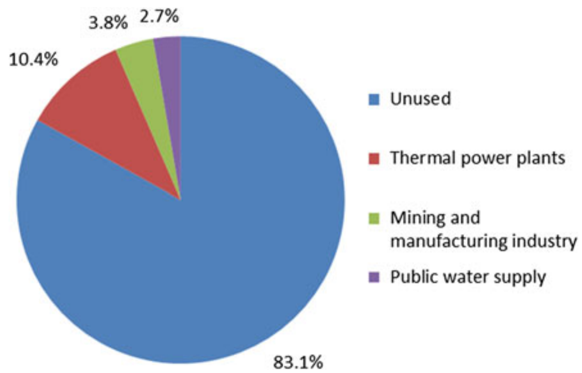

Fig. 12.3 Available water resources and water use in Germany (2007) (Reproduced from ATT et al. 2011)

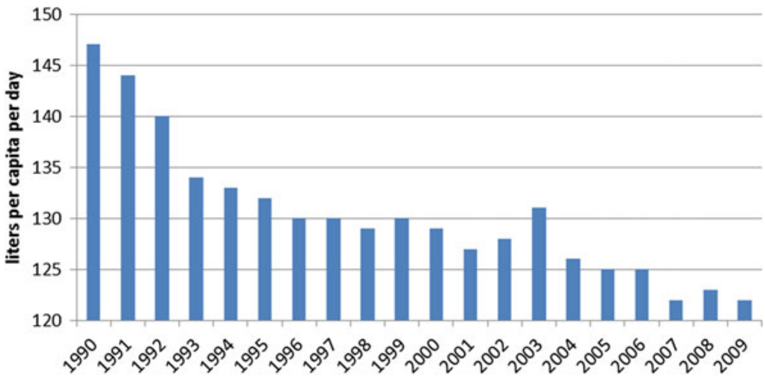

Fig. 12.4 Water consumption in Germany (1990–2009) (Reproduced from ATT et al. 2011)

consumption and to the amount of pollution attributable to water users, (c) installation of water meters and water-saving sanitary facilities, as well as the use of water-saving household appliances and fittings, and (d) better consumer awareness (Althoff 2007). Moreover, the water supply utilities are permanently in contact with their customers through their customer and information centers to inform their customers about responsible water use, water quality, and pricing. Furthermore, increasing use of rainwater for watering the garden and toilet flushing, and for agricultural use through drip irrigation, has also reduced the need for treated water. In the industrial sector, the reasons for the decrease in water consumption are (a) the application of resource-friendly production processes, (b) a decrease in water purchases, and (c) an increase of its own water production. In Germany, industry makes up 96 percent of its water demand by its own production (Althoff 2007).

12.4 Development of Wastewater Treatment in Germany

12.4.1 The History of Wastewater Treatment in Germany

With rapid industrialisation and urbanization in Germany, wastewater has been a concern since the second half of the nineteenth century (Seeger 1999). In order to meet the requirements of wastewater treatment, the development of wastewater treatment technology in Germany has made steady progress. It started in the late nineteenth century with mechanical–biological treatment applications in agricultural fields (Seeger 1999). Before the First World War, sludge digestion in special digestion tanks became the standard for the larger urban sewage plants, and then artificial biological treatment with high-loaded trickling filter was introduced into some sewage plants. After the end of the Second World War, full biological treatment became the main goal for urban wastewater treatment, and the activated sludge process became the dominant method of treatment in the 1950s.

When the Federal Water Act came into force in 1957, the first common framework for water protection was established in Germany (Seeger 1999). However, wastewater treatment in Germany faced an excess of sludge due to urban extension. It meant that “huge amounts of sludge forced the abandonment of the dewatering of the digested sludge on drying beds” (Seeger 1999). Accordingly, artificial sludge dewatering was added to the sewage plants during the 1960s. In 1979, the First General Regulations Concerning the Discharge of Municipal Wastewater set up target values for the parameters BOD (Biological Oxygen Demand) and COD (Chemical Oxygen Demand). In the 1980s, the method for tertiary wastewater treatment was developed, and was put into operation. For example, the first municipal sewage plant in Berlin was able to conduct denitrification to remove nitrogen compounds during this period (Pöpel et al. 1997). As a statutory requirement for tertiary wastewater treatment, the amendment of the First General Regulations Concerning the Discharge of Municipal Wastewater issued target values for nitrogen and phosphorus in 1989. Consequently, the requirement of nutrient removal for all municipal sewage plants became the final step in current wastewater treatment.

12.4.2 Current Wastewater Treatment in Germany

The goal of water protection in Germany is to conserve or restore unimpaired ecological functioning of all water bodies. The discharge of untreated wastewater into rivers and lakes is not permitted in Germany. According to Article 57 of the *Federal Water Act*, “discharges of wastewater into water bodies is only permissible if the pollution load of the wastewater is kept to the lowest level achievable by means of the best technology available” (BMU 2014). The Federal Ministry for the Environment states that “Germany is the European country with the highest wastewater reprocessing and recycling rate.” Furthermore, according to a 2011 Profile of the German Water Sector, more than 77 percent of customers were extremely satisfied or satisfied with the services of their wastewater disposal utilities in 2007, indicating that the overall satisfaction of customers with wastewater disposal remained high.

12.4.2.1 Wastewater Treatment in Public Sewage Plants

Currently, there are more than 6,900 municipal wastewater disposal companies and almost 10,000 wastewater treatment plants in Germany. The municipal sewage plants are largely located in small population areas, with approximately 55 percent of these plants serving fewer than 5,000 people and merely 0.4 percent serving more than 100,000 people (see Fig. 12.5). A total of 78 million inhabitants are connected to centralized municipal sewage plants (Umweltbundesamt, the German Federal Environment Agency 2014).

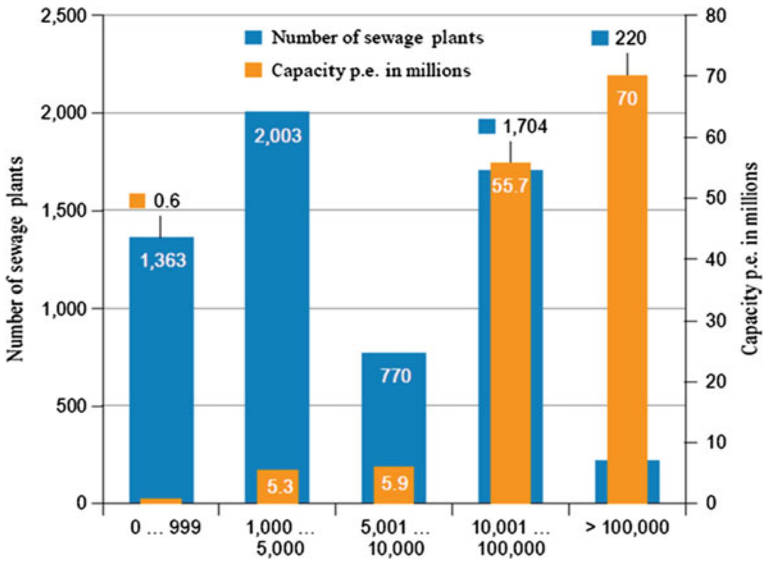

Fig. 12.5 The capacities of sewage plants (BMU 2011)

A total of 10 billion cubic meters of wastewater (i.e. sewage water, rainwater, and infiltration water) was treated in the public sewage plants in 2010 (Umweltbundesamt, German Federal Environment Agency 2014). Of this, only 0.03 percent was not treated by a biological wastewater treatment process (see Fig. 12.6). With the implementation of Annex 1 of the *Waste Water Ordinance* and *EU Urban Waste Water Treatment Directive,* as well as the development of wastewater treatment over the last decades in Germany, by 2010, 98.1 percent of municipal mechanical–biological plants have the capacity to remove nitrogen and phosphate, which has brought a significant improvement in biological water quality.

Furthermore, from 2002 to 2011, the share of wastewater treated in biological sewage plants with selective removal of nutrients increased to 82 percent. As a consequence, in 2011, on average the municipal wastewater treatment plants achieved a reduction in nutrient loads of 91 percent for phosphorus and 81 percent for nitrogen, which clearly exceeded the requirements of the EU Urban Waste Water Treatment Directive (Directive 91/271/EEC) (Umweltbundesamt, German Federal Environment Agency 2014). This is a reduction of 75 percent for both substances taken together.

The 98 percent biological treatment is a high standard and one would expect that most of the contaminants, including pharmaceuticals, pesticides, and personal care products (PPCPs), would be removed or oxidized. The most recent (July 2014) information we have is that where surface water is used for drinking water, greater efforts are made to clean up the wastewater. For example, in North Rhine and Westfalia, wastewater treatment now includes charcoal filtration, membrane filtration, and ozonation. Nevertheless, it would be good to see results of tests that show the efficacy of the treatment and its effects on levels of PPCPs in Germany (see Sect. 12.5).

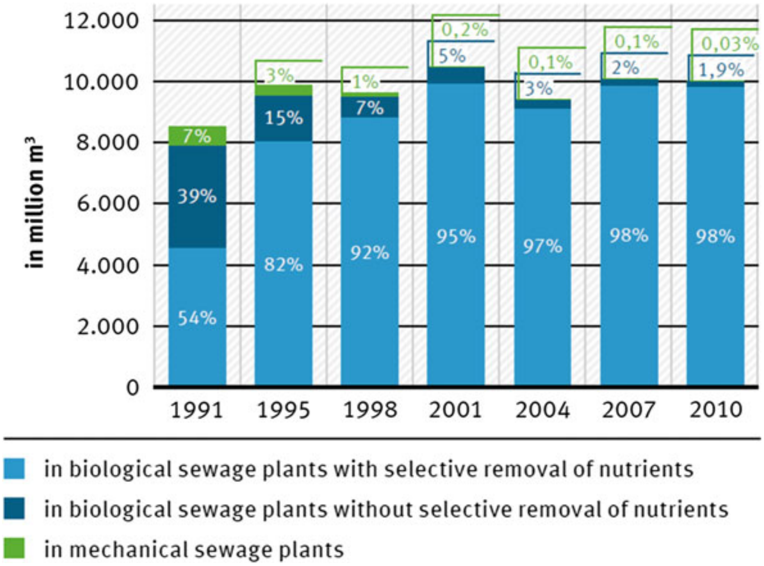

Fig. 12.6 Wastewater volumes treated in public sewage plants (Umweltbundesamt, German Federal Environment Agency 2014)

12.4.2.2 Security of Sewage Network

In 2004, 95 percent of sewage network operators documented that they had completed inspection (Althoff 2007). The results from the inspection indicated that minor damages existed for approximately 21 percent of the sewage network, which needed to be rehabilitated in the long-term, while around 20 percent of the public sewage systems were in need of rehabilitation in short or medium term (Althoff 2007). Moreover, almost one third of the existing sewers had been reconstructed over the last decade. This means that the service life of these sewers will be extended (Althoff 2007).

Althoff pointed out that “a major factor for long-term disposal security is the continuous investment in maintenance and renewal of the infrastructure.” In Germany, the average investment per cubic meter of wastewater was 1.27 euros, followed by England/Wales with 0.91 euros, France with 0.72 euros, and Italy with 0.11 euros from 1995 to 2003 (VEWA-Studie 2006, as cited in Althoff 2007). In 2005, the water and wastewater utilities invested about 8 billion euros on sewage networks. It should be noted that all investment costs are financed through prices and charges, while in other countries investments are financed partially by the municipalities.

12.5 Micropollutants in Three Countries

As mentioned above, the presence of pharmaceuticals and personal care products (PPCPs) in wastewater has become a global issue. From 1996 to 1998, a comprehensive study in Germany was undertaken by Ternes (1998, 2000). He detected 32 of 55 pharmaceuticals, 4 of 6 hormones, 5 of 9 metabolites, and 5 of 6 biocides in the outflow of 49 wastewater treatment plants. He further found that "the receiving water bodies contained concentrations of beta-blockers and antiepileptic agents in excess of 1 μg/L" (Ternes 2000). However, Ternes et al. (2004) pointed out "these comprehensive monitoring studies[3] and the many subsequent individual studies (Heberer 2002; McArdell et al. 2003; Huang; Sedlak 2001, Metcalfe et al. 2003; Boxall et al. 2004; Jones et al. 2001) included only a small subset (less than 15 percent) of the pharmaceuticals and personal care products (PPCPs) predicted to potentially enter the environment." They further noted that "studies were launched to investigate the effects of individual PPCPs on biota" (Huggett et al. 2002; Ferrari et al. 2003; Wollenberger et al. 2000). However, because of incomplete data, researchers still lack a complete understanding of the environmental effects of most PPCPs. Thus, no one knows whether the relatively low environmental concentrations of PPCPs produce adverse effects on aquatic and terrestrial biota or whether the toxicity of complex mixtures might be totally different from that of individual compounds (Altenburger et al. 2000). Therefore, Ternes et al. (2004) recommended that "for precautionary reasons, we can be proactive and reduce the inputs of micropollutants to the environment as completely as possible through the introduction of cost-effective control options".

One very cost-effective way of neutralizing PPCPs is through a process of biological degradation, which usually requires a few days of "retention time" of the sludge. The process of biological degradation of the contaminants can be described as follows (Ternes et al. 2004). In wastewater, PPCPs occur primarily at concentrations of less than 10^{-4} g/L. At these levels, biological transformation or degradation of the trace pollutants occurs only if a primary substrate is available so that the bacteria can grow on it. In this way, cometabolism occurs, in which case the bacteria break down or partially convert the trace pollutant. Alternatively, mixed-substrate growth takes place and the bacteria use the trace pollutant as a carbon and energy source and may mineralize it totally. Thus biological degradation is of crucial importance and adequate time must be allowed for the degradation to take place.

Moreover, in order to comply with the requirements of the EU Urban Waste Water Treatment Directive[4] for discharges of wastewater treatment plants to water bodies such as the coastal waters of the North Sea, Ternes et al. (2004) proposed a time period of 12 to 15 days of retention time for adequate degradation of pollutants

[3] These studies were carried out in the UK, Canada, and USA.

[4] For example, as we referred in Sect. 12.2.1, the EU Urban Waste Water Treatment Directive requires a reduction of 75 percent for nitrogen; Germany exceeds this reduction requirement.

in medium and larger sized sewage plants, and that some pollutants (i.e. Carbamazepine and diazepam) might require an even longer period, where specific information on pollutants is tested or known.[5] However, Ternes et al. (2004) noted that "many wastewater treatment plants in the United States and the EU do not operate with solid retention times long enough to satisfy these requirements." But of course Germany is way ahead of most countries in total tertiary treatment of all wastewater, as noted above.

Furthermore, since many PPCPs have limited biological degradability, these compounds cannot be totally removed by wastewater treatment plants before entering into water bodies (Ternes et al. 2004). Treatments such as Nanofiltration (NF) or activated carbon adsorption are more cost-effective, and therefore they are applied to groundwater recharging or directly for water that would be reused for drinking water (Ternes et al. 2004). Compared to these short-term solutions, source control or emission control appears to be the permanent, cost-effective measure for most compounds (Ternes et al. 2004). The other more sustainable measures that can be implemented are as follows (Ternes et al. 2004):

a. Since hospital wastewater contains pharmaceuticals and antibiotic-resistant bacteria, it should be treated separately by using a membrane bioreactor followed by ozonation of the effluent. These applications could also be beneficial to the hospital when the treated wastewater is reused for flushing toilets and for gardening or discharged directly; such measures would aid conservation and also reduce associated drinking water fees as well as wastewater fees.
b. Providing consumers with information on the environmental impacts of PPCPs would result in a significant reduction in the disposal of these substances into household wastewater.
c. There could be direct control of the disposal of PPCPs; for example, expired products could be separated and sent directly for incineration at wastewater treatment plants.
d. Separation and segregated treatment of urine would significantly reduce the loading of wastewater because pharmaceuticals are excreted to a great extent in urine (Larsen and Gujer 1996; Klaschka et al. 2003).

However, these measures require political decisions, financial support, and public awareness, and they may not be achieved in a short time. In Germany, micropollutants continue to be a key issue in the wastewater sector. As noted by the German Federal Ministry for the Environment (BMU 2014), "a major challenge for the future will be the elimination of pollutants in wastewater which, to date, have not been taken into account, such as pharmaceutical residues, antibiotics from animal husbandry or chemicals displaying hormone-like effects even in minute

[5] The lipid regulator bezafibrate, the antibiotic sulfamethoxazole, and the antiphlogistics ibuprofen and acetylsalicylic acid require a sludge age of 2–5 days for significant degradation; 17 -ethinylestradiol, the anti-inflammatory diclofenac, the contrast medium iopromide, and the antibiotic roxithromycin need 5–15 days. Carbamazepine and diazepam are not degraded even at a sludge age >20 days (Ternes et al. 2004).

quantities. Current treatment technologies are not able to remove these trace substances. There are first tentative technologies such as special membranes or oxidization processes which make removal possible. However, to date there are no legal thresholds which could serve as a guidance for sewage plant operators."

It is not quite true to say that treatment technologies are not available to treat or oxidize PPCPs. UV-based Advanced Oxidation equipment exists and has been shown to be a very effective method of treating PPCPs (Trojan Technologies 2000) (see information on the Trojan case studies in Dore 2015, Chap. 4, "Water Policy In Ontario"). These Advanced Oxidation processes can be used for treating wastewater or for drinking water. These techniques are also very cost-effective.

12.5.1 Micropollutants in the Netherlands

There are now over five million man-made chemicals, approximately 100,000 of which are defined in the European Inventory of Existing Commercial Chemical Substances (EINECS) list (Van Leeuwen et al. 2007). These chemicals are increasingly evident in the environment everywhere, including the high Arctic. There is also growing evidence of the adverse health impacts these chemicals are having on many life forms, especially aquatic life.

Some of the adverse impacts are due to improper disposal of pharmaceuticals and personal care products (PPCPs) into source waters that are also sources of drinking water. Many compounds have been detected in drinking water sources (Richardson 2007 and Loos et al. 2009). For example, in the Netherlands, over 1,300 compounds have been detected in drinking water sources since 1983, especially pharmaceuticals such as analgesics, antibiotics, anti-epileptics, and X-ray contrast media. Some of them are in very low concentrations, as scientific detection methods keep advancing so that lower and lower concentrations can be detected. For some of these compounds, the concentrations are below the level that will threaten human health (i.e. 10–170 ng/L) (Van Genderen et al. 2000; Stan et al. 1994; Zuccato et al. 2000; Ternes 2001; WHO 2011; Christensen 1998; Schulman et al. 2002; Webb 2001; Mons et al. 2003; Mons 2003; Versteegh and Dik 2007).

Table 12.1 presents an overview of concentrations of some of the pharmaceuticals detected in treated water in the Netherlands, in comparison with their safe drinking water levels (SDWLs) and their minimum therapeutic doses. The concentrations are all far below the SDWLs. Furthermore, lifetime consumption of this drinking water would result in a total accumulated dose (I_{70}) of less than one daily dose for therapeutic treatment.

Therapeutic health effects are therefore not to be expected, even after chronic exposure, let alone toxic health effects (which usually occur at higher doses than therapeutic effects). Nevertheless, the presence of such pharmaceuticals (Ter Laak et al. 2010; De Jongh et al. 2012) and drugs of abuse (De Voogt et al. 2011) receives a lot of negative media attention and may have a negative effect on consumer confidence in the quality of drinking water.

Table 12.1 Concentrations of some of the pharmaceuticals detected in treated water in The Netherlands in comparison with safe drinking water levels (SDWL) and I_{70} values (Mons et al. 2013)

Compound	MCLs observed in treated drinking water (ng/L)	DWL[a] (ng/L)	DWL Daily drinking water consumption needed to reach DWL	I_{70} value (mg)[b]	Therapeutic dose (mg/day)	I_{70}/therapeutic dose (percent)
Acetylsalicylic acid	122	25 × 103	205 L	6.2	20	30
Diclofenac	18	7,500c	417 L	0.9	15	6
Carbamazepine	90	50 × 103[c]	556 L	4.6	100	5
Prozac (fluoxetine)	10	10,000[c]	1000 L	0.5	20	2.5
Bezafibrate	20	35,000[c]	1750 L	1	67	1.5
Metoprolol	26	50,000[c]	1923 L	1.3	100	1.3
Fenofibrate	21	50,000[c]	2381 L	1.1	100	1.1
Clofibric acid	136	30,000[c]	221 L	6.9	1200	0.6
Phenazone	29	125,000[c]	4310 L	1.5	250	0.6
Ibuprofen	28	150 × 103[c]	5357 L	1.4	300	0.5
Paracetamol	33	150,000	4545 L	1.7	1200	0.15
Lincomycine	21	30 × 103	1429 L	1.1	1200	0.1
Sulfamethoxazole	40	75 × 103	1875 L	2	2000	0.1
Amidotrizoic acid	83	250 × 106[d]	3 × 106 L	4.2	50,000d	0.008
Iopamidol	68	415 × 106[d]	6 × 106 L	3.5	83,000d	0.004
Iopromide	36	250 × 106[d]	7 × 106 L	1.8	50,000d	0.004
Iohexol	57	375 × 106[d]	7 × 106 L	2.9	75,000d	0.004

[a] SDWL: safe drinking water level, based on either acceptable daily intake or maximum residue limit
[b] I_{70} value: amount ingested after 70 years of consumption of 2 L of drinking water per day, with the maximum concentration of the pharmaceutical observed in drinking water
[c] Provisional SDWL, based on lowest therapeutic dose and uncertainty factor of 100
[d] x-ray contrast medium. The highest dose used is assumed to have no effect

In order to assure drinking water safety, the EU Drinking Water Directive (1998) has set a number of microbiological, chemical, and organoleptic parametric standards, and required drinking water in all member states to meet these minimum requirements.[6] The Netherlands has regulated its own MCLs for a number of parameters based on the *Directive*, some of which are even lower than those in the *Directive* (Versteegh and Dik 2007) . Currently, the Netherlands has reached a high quality in drinking water by using advanced water treatment technologies and frequent water quality monitoring. As a result, bottled water consumption in the Netherlands is the lowest among the EU Countries (The Dutch association of soft drinks, waters, and juices 2009 and Geudens 2012). However, many emerging contaminants have not been regulated in the Netherlands as well as in the other EU countries, since toxicological information[7] for these contaminants (or compounds) is unknown. Accordingly, the Dutch water utilities have developed a new approach called Q21 ("Drinking Water Quality for the twenty-first Century") (Van Der Kooij et al. 2010). To achieve an impeccable drinking water quality, as a part of the Q21, target values (i.e. acceptable/tolerable concentrations) for those emerging contaminants have been proposed as an addition to the regulatory standards (Mons et al. 2013). The derivation of these target values is mainly based on the approach called *Threshold of Toxicological Concern* (TTC), (Mons et al. 2013).

12.5.1.1 The Threshold of Toxicological Concern (TTC)

Frawley (1967), Rulis et al. (1989), and Munro (1990) examined 217 carcinogens and found only a small chance (4 percent) that a new chemical would contribute a higher risk for cancer. In a later paper, Munro et al. (1996) found higher thresholds for 613 compounds tested for toxicity endpoints other than carcinogenicity. They further divided these compounds into three structural "Cramer classes." Higher TTC values, up to 1800 mg per person per day are assigned to Cramer class I, implying these compounds have significant toxicity, namely substances of high concern, while the lower values (90 mg per person per day) are put in Cramer class III, showing simple chemical structures with efficient modes of metabolism (i.e. substances of low concern). Substances of *in-between* concern are classified in Cramer class II (Cramer et al. 1976). As a special case, organophosphates have a lower TTC, i.e. 18 mg per person per day, which is below 90 mg per person per day (or class III). Also recent reviews on low-dose compounds found effects at doses far below those related to the Cramer class III (Macon et al. 2011 and Andrade et al. 2006). Thus, a TTC of 0.15 mg per person per day has been determined as a

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[6] As the EU Drinking Water Directive (1998) noted, "for the purposes of the minimum requirements of this Directive, water intended for human consumption shall be wholesome and clean if it is free from any micro-organisms and parasites and from any substances which, in numbers or concentrations, constitute a potential danger to human health".

[7] Toxicological information reveals that the emerging substance is present in drinking water at a concentration below the level that will threaten human health (Mons et al. 2013).

threshold of concern. The probability of human intake at this threshold level (i.e. TTC of 0.15 mg per person per day) is 86 to 97 percent, but the cancer risk would be less than 1×10^{-6} (or 0.000001) (Kroes et al. 2004). Mons et al. (2013) noted that although the TTCs of compounds below the Cramer class III were observed frequently, effects observed at doses below those related to the TTC of 1.5 mg per day per person have not been reported. Therefore, "this value appears to be sufficiently protective and it is regularly reviewed with the latest knowledge and data" (Mons et al. 2013).

However, it should be noted that according to the European Food Safety Authority (2012), the TTC approach is not to be applied to (a) high potency carcinogens (i.e. aflatoxin-like, azoxy- or N-nitroso-compounds, benzidines, hydrazines), (b) inorganic substances, (c) metals and organometallics, (d) proteins, (e) steroids, (f) substances that are known or predicted to bioaccumulate, and (g) nanomaterials. These must be treated separately as they are dangerous to human health.

12.5.1.2 Target Values

Using the TTC approach, the target values for micropollutants which are anthropogenic drinking water contaminants, can be determined. The proposed target values are summarized in Table 12.2. The target values for individual genotoxic and steroid endocrine chemicals were set at 0.01 mg/L. For all other organic chemicals the target values were set at 0.1 mg/L. The target values for the total sum of genotoxic chemicals, the total sum of steroid hormones, and the total sum of all other organic compounds were set at 0.01, 0.01, and 1.0 mg/L, respectively. However, there are two concerns about target values for organic contaminants in drinking water. First, "many different compounds can be present individually in drinking water at concentrations below the target values, but as a mixture, the sum of all compounds together might still threaten human health" (European Commission 2009). Therefore, there is justification for setting a target value for the *sum* of all contaminants, "…to avoid the presence of a wide variety of compounds at levels just below their individual target value" (Mons et al. 2013). Moreover,

Table 12.2 Proposed target values for organic contaminants in drinking water in the Netherlands (Mons et al. 2013)

Compound group	Target value (mg/L)
Single genotoxic organic chemicals	0.01
Single (synthetic) steroid hormones	0.01
All other single organic chemicals	0.1
Total sum of genotoxic compounds	0.01
Total sum of (synthetic) steroid hormones	0.01
Total sum of all other organic chemicals	1

"concentration action" can lead to toxic effects of the total mixture, while the "individual action" does not result in toxic effects of the mixture. However, individual action with different modes can lead to higher toxic effects of a chemical mixture than the effects of an individual chemical (European Commission 2009). Therefore, from a precautionary point of view, some emphasis must be placed on "concentration action," as this would enhance safety (European Commission 2009). As an example, in order to capture the cumulative risk of pesticides, the US EPA has set both a common toxic effect of individual pesticides as well as the toxic effect of a mixture of pesticides, called "concentration action" (USEPA 2002). But in general, this approach is not feasible for many other chemicals and their compounds, as knowledge of the toxic effects of many compounds of chemicals found in water is not yet available (Mons et al. 2013).

12.5.2 Micropollutants in the USA

A study conducted in the USA by Kolpin et al. (2002) examined concentrations of 95 organic compounds (i.e. pharmaceuticals, antioxidants, phytosterols, biocides, and flame retardants) in water samples from a network of 139 streams across 30 states between 1999 and 2000. They detected 82 of the 95 compounds in at least one stream sample. To ensure drinking water safety, the USA has already regulated Maximum Contaminant Levels for a number of chemical, microbiological, and radiological parameters under the *Safe Drinking Water Act* (USEPA 1996). Moreover, the US Environmental Protection Agency has released a *Candidate Contaminant List* (CCL) in which the contaminants are known to be in drinking water sources and should be regulated in the future. The drinking water utilities are responsible for monitoring a number of unregulated and emerging contaminants periodically under the *Unregulated Contaminant Monitoring Rule* (UCMR). The new compounds are needed to determine whether they should be regulated and included in the *list* based on the monitoring results. According to US EPA, the first Candidate Contaminant List was issued in 1998 and the latest list was released in 2009 (USEPA 2009).

In the USA, there are no general target values used for the unregulated contaminants, but specific health-based target values have been set for each contaminant. As compared with the Netherlands, Mons et al. (2013) pointed out that "the Dutch Q21 approach differs from the US approach with the Contaminant Candidate List where information on adverse health effects is essential in deciding whether or not to regulate the specific compound." Furthermore, the Candidate Contaminant List is a "substance-specific approach" which needs to be conducted for each individual contaminant, while the target values, as a part of Q21 approach, can be derived and applied to all substances (Mons et al. 2013). In addition, the US approach "does not provide guidance for situations where compounds are detected in drinking water in concentrations below toxicological standards" (Mons et al. 2013).

12.5.3 Micropollutants in Germany

The German Federal Environment Agency has also developed recommendations for those micro- pollutants, which are more or less equivalent to "thresholds of toxicological concern" although the micropollutants are not regulated so far in Germany. Instead, depending on the amount of toxicological information available for specific substances, Germany has set what are called "Health-Related Indicator Values (HRIV)", which range from 0.01 to 3.0 mg/L. A Health-Related Indicator Value of 0.1 mg/L has been set as a precautionary value, which should allow lifelong consumption of the drinking water for 70 years (Umweltbundesamt, German Federal Environment Agency 2003, as cited in Mons et al. 2013). The value of 0.1 mg/L applies to both nongenotoxic compounds and the majority of genotoxic compounds, while highly genotoxic compounds cannot be used for lifetime exposure, but are safe for short periods only (Umweltbundesamt, German Federal Environment Agency 2003, as cited in Mons et al. 2013) . Table 12.3 shows maximum values for lifelong exposure to unregulated contaminants in drinking water in Germany, in which the Health-Related Indicator Values can be up to, or even over 3 mg/L, depending on the quality of the available information.

As a matter of fact, the higher Health-Related Indicator Values can be applied for chemicals if toxicological data shows sufficient safety (Mons et al. 2013). It should be noted that the Health-Related Indicator Values "only consider the prevention of adverse health effects," and "not the principle that anthropogenic contaminants do not belong in drinking water." The shortcoming of the German approach is that the sum of values of chemical compounds from *mixtures of contaminants* in drinking water has not been used. In other words, something comparable to the Dutch concern for "concentration action" of mixtures of compounds should be adopted in

Table 12.3 Maximum values for lifelong exposure to unregulated contaminants in drinking water in Germany (Umweltbundesamt, German Federal Environment Agency 2003, as cited in Mons et al. 2013)

HRIV (mg/L)	Explanation
0.1	No toxicological data available
0.3	Only genotoxicity data available, indicating the substance to be nongenotoxic
	No other toxicological data available
1	Substance proven nongenotoxic (see above). Data on neurotoxicity and germ cell damaging potential available, indicating a value <0.3 mg/L
3	Substance neither genotoxic nor germ cell damaging nor neurotoxic
	In vivo data on subchronic oral toxicity available, indicating a value lower <1 mg/L
>3	At least one chronic oral study is available enabling (almost) complete toxicological information and not indicating a value <3 mg/L

Germany. Mons et al. (2013) pointed out that the "presence of a range of [individual] contaminants at concentrations just below their individual target value is undesirable [by itself], because it demonstrates that a variety of [mixtures of] contaminants can pass drinking water treatment". Hence the total mixture should also be a serious concern, as it is in the Netherlands.

Although the drinking water treatment plants in Germany are not able to remove all micropollutants in the main cites, "drinking water conditioning in Germany aims at removing pollutants (also micro-pollutions) from water to such a degree, that there is no risk for human health [even if there is] … lifelong consumption of the drinking water (2 liters daily for a period of 70 years)" (Markard 2014). As Germany is highly industrialized and densely populated, it is not surprising if micropollutants are detected in drinking water samples. Thus, the German government attempts to "keep [a] hazardous substance which can influence drinking water quality, as low as achievable according to the generally acknowledged technical standard of treatment within [reasonable] expenditure [limits]" according to the "minimization rule" of the German Drinking Water Ordinance (Markard 2014).

At this point it is worth recalling what was noted above on the high quality of wastewater treatment in Germany. Biological degradation of wastewater is practised on a vast scale, with only 0.03 percent of wastewater that is not subjected to biological treatment (Fig. 12.6). If we assume that the wastewater treatment plants use the scientifically required time for biological degradation, then we can expect that in Germany, PPCPs are well below the Health-Related Indicator Values (HRIV) stated in Table 12.3, which are themselves quite stringent. Although the HRIV are above the targets set in the Netherlands (see Table 12.1), they are still below the I_{70} limit, which are quantities *ingested after 70 years of consumption of 2 L of drinking water per day, with the maximum concentration of the pharmaceutical observed in drinking water.*

Furthermore, note that the PPCPs being well below the Health-Related Indicator Values are only relevant for the portion of the population that relies on surface water, which may have been subject to wastewater discharges. As noted above, only 24 percent of the drinking water comes from surface water, and the rest from groundwater, which is presumably free of all micropollutants. Therefore, we can conclude that in Germany, *treated* drinking water is of the highest quality, probably better than Class 6, in terms of the classification put forward in Chap. 3.

12.6 Cost Structure of Water Supply and Wastewater Discharge

12.6.1 Water Supply

As shown in Chap. 4, the EU Commission recommends a three-part tariff that includes (a) a fixed component, to cover the fixed financial costs of supply, (b) a charge per unit of water used, and (c) a charge per unit of pollution produced.

Germany complies strictly with this structure of water prices. Moreover, Germany also maintains transparency in pricing and the corresponding disclosure of information to the consumer as proposed in the EU announcement (2007, July) (Althoff 2007). Consumers are informed about the price level and structure through official announcements, local and regional press, mail circulars, water invoices, and so on (Althoff 2007).

The fixing of prices and charges is subject to strict statutory regulation and the pricing is controlled by several governmental levels (Althoff 2007). Municipal utilities of water supply and wastewater disposal are controlled at the municipal level by the town or local council and within the associations by the respective bodies and by the municipal supervisory authority (Althoff 2007). The municipal supervisory authority is responsible for implementing local tax laws. Moreover, pricing is controlled by representatives of the current board of supply and disposal utilities, who are democratically elected. Furthermore, both public and private water supply utilities are regulated by the regional associations, and the utilities invoice their services directly by price to the consumers (Kappel and Schmidt 2007, as cited in Althoff 2007).

In Germany, pricing is based on five general principles, which were published in a 1982 water supply report by the federal government, and were also included in the local tax laws of the federal Länder (Althoff 2007). These principles can be summarized as follows (Althoff 2007):

a. Cost recovery principle. All costs associated with water supply or wastewater disposal are to be covered by the price or charge. The cost recovery principle makes the long-term security of the water supply and wastewater disposal possible since all investment costs for construction and maintenance are included in prices and charges.
b. Breakdown of charges by consumer groups, according to the costs attributable to specific consumer groups. For example, although an industrial customer may use the same amount of water as thousands of households, the distribution costs that the utilities incur are very different.
c. Considerations of cost structure for the determination of the base price and the quantity price. The average costs for water supply consist of fixed costs (70–80 percent) and variable costs (20–30 percent) (see Fig. 12.7). Fixed costs include the costs for maintenance of facilities, which are incurred independently of the system's usage rate, while variable costs are determined by the quantity of water produced, including costs of wear and tear of equipment for water treatment, electricity costs for pumping, water abstraction charges, and so on. In roughly 95 percent of German water supply utilities, price can be divided into a fixed annual base price and a quantity-dependent price (euros per cubic meter). In fact the fixed annual base price only makes up 10 percent of the total price, and therefore the structure of the water prices is dominated by variable costs.

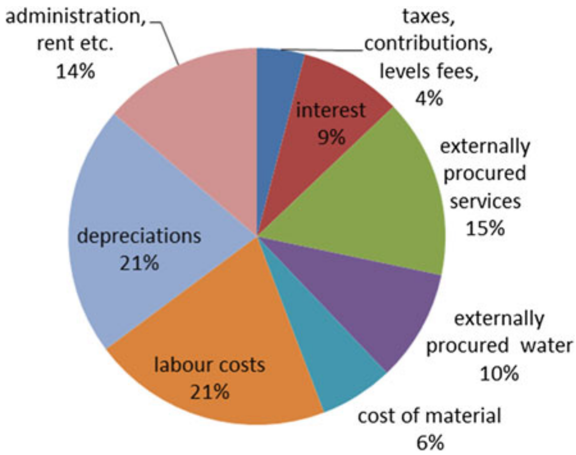

Fig. 12.7 Cost structure in water supply in 2004 (Althoff 2007)

d. Adequate interest yield for equity capital and debt capital. In order to ensure long-lasting security of water supply, water supply utilities should obtain a profit that allows at least a current market rate of interest on the capital invested.[8]
e. Adequate maintenance of the capital infrastructure. For long-term security of supply, there must be adequate provision for the maintenance and renewal of capital equipment. The municipal water supply system generally has a long service life of 20 years and up to 70 years for pipelines and hydrants. Moreover, after the service life, the replacement of equipment must be based on expected future prices and not the historical prices of the old equipment.

12.6.2 Wastewater Disposal

In Germany, the *freshwater standard* and the *split charge standard* are two ways of calculating wastewater charges. The main difference between the two methods is that the cost for rainwater collected from public property is separated and its cost is divided equally over the community. On the other hand the cost of rainwater collected from private property is charged to the owner of the property.

The *freshwater standard* assumes that the amount of freshwater used by an owner ends up eventually as wastewater. So that is the first component of the total wastewater charge paid for by the private owner. The second component is then based on the amount of rainwater collected and processed from streets and from public property, which is called the split charge. Most large municipalities are able

[8] This corresponds with the principles of economic activity of municipalities that are stated in the municipal bylaws of the federal states (Althoff 2007).

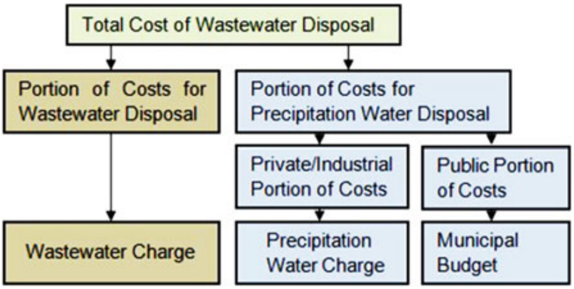

Fig. 12.8 Split wastewater charge (Althoff 2007)

to use the split charge method whereas this may not be possible for smaller municipalities (see Fig. 12.8).

When consumers pay according to the *split charge standard*, they receive an invoice, on which wastewater and rainwater are listed separately. In particular, charges for water due to precipitation are calculated on the basis of square meters of paved and drained plot area. This method of charging for wastewater is designed to maintain fairness, as the amount of water based on the polluter pays principle refers only to the actual water used and discharged by a consumer, who is not burdened with a charge for rainwater that falls on the streets. The existence of a fair and ecological wastewater charge is the justification for the separation of the *freshwater standard* and the *split charge standard*. In North Rhine-Westphalia, the *split charge standard* has been implemented since May 2008, in accordance with the decision of the Federal Administration Court (Althoff 2007). As stated previously, the *split charge standard* is used in many municipalities with a higher density of population.

Fixed and variable costs include maintenance of wastewater infrastructure including depreciation, interest, labor costs, and other costs (see Fig. 12.9). Approximately 75–85 percent of the total costs for wastewater disposal are fixed costs, which do not depend on the amount of wastewater collected and treated by a wastewater treatment plant. Thus fixed costs in Germany are the dominant cost component (Althoff 2007).

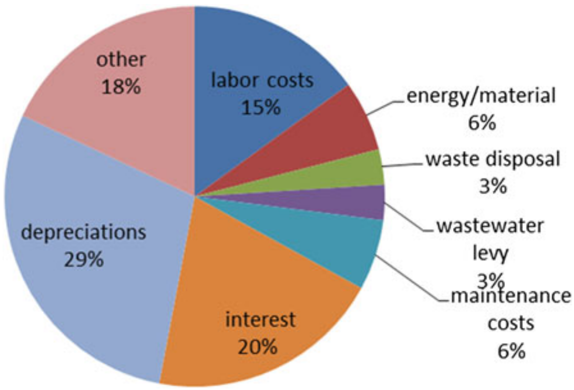

Fig. 12.9 Cost structure in wastewater disposal in 2005 (Althoff 2007)

Wastewater infrastructure lasts a long time and is amortized over the entire service life. The long service life of capital-intensive technical facilities is a particular challenge for the wastewater disposal utilities when it comes to proper accounting. The service life of a sewage system is approximately 50 to 80 years. Wastewater charges follow the cost recovery principle, and therefore "the consumer pays the costs which arise for the wastewater collection and wastewater treatment" (Althoff 2007). Althoff further pointed out that "these costs also include the wastewater tax, which is a statutory extra fee. "The amount of the wastewater tax depends on the residual contents of wastewater substances in the discharged wastewater. This was designed as a "polluter pay tax" on the residual contents of the wastewater, but as these contents cannot be traced to the particular consumer who discharges these pollutants, the tax has lost its "steering effect and function" (Althoff 2007). This is a common problem with all nonpoint source pollution, as the source cannot be identified.

12.7 Mean Water Price in Germany

12.7.1 Fiscal Framework

In Germany, some water utilities are taxed like other private corporations. Water supply utilities are taxed with a uniform and reduced turnover tax rate of 7 percent. The public wastewater utilities as "sovereign undertakings" are exempted from corporate tax and turnover tax, while the private wastewater utilities are subject to a full turnover tax rate of 19 percent with the usual deductible costs (Althoff 2007).

12.7.2 Drinking Water

In 2002, the average drinking water price in Germany was 1.80 euros per cubic meter. In Italy and Spain, even though droughts occur frequently, the average drinking water prices are much lower: 0.73 and 0.72 euros per cubic meter, respectively. The low water prices in Italy and Spain could be indirect subsidies for agriculture. In Great Britain and France, the average drinking water prices are 1.25 and 1.09 euros per cubic meter, respectively. Between 2000 and 2009, the German drinking water price index increased by 5 percent though this was below the inflation rate, while the general price increase amounted to 15.9 percent during the same period (see Fig. 12.10). Finally note that, in Germany, the average share of households' expenditures on water bills is only 0.5 percent (German Federal Association of the German Gas & Water Industries (BGW) 2004 and German Federal Association of Energy and Water Industries (BDEW) 2007, as cited in Althoff 2007).

Due to regional differences in water resources, treatment and distribution costs, water prices in Germany vary for different regions (BGW 2004 and BDEW 2007,

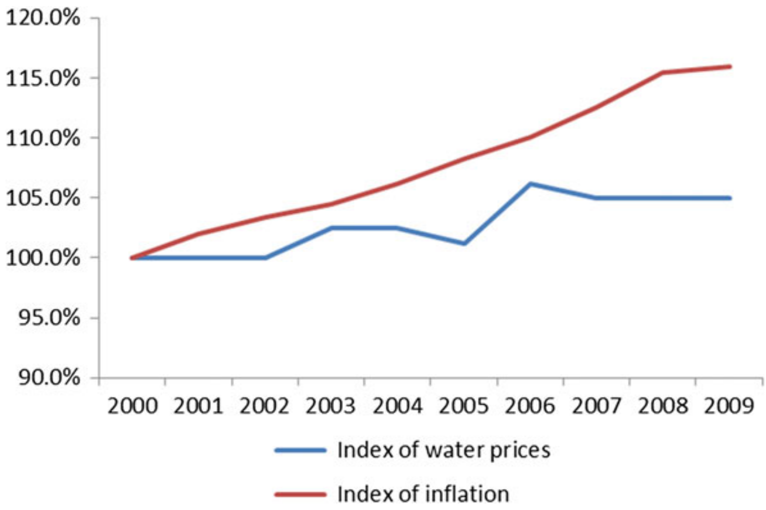

Fig. 12.10 The drinking water price index compared to inflation from 2000 to 2009 (Reproduced from ATT et al. 2011)

as cited in Althoff 2007). Factors that affect the costs include (a) the distance from the area of abstraction to the consumers, (b) the quality of raw water and the costs for the treatment of drinking water, (c) the connection density of the households and enterprises provided, and topographic differences that affect the cost of providing water pipes, (d) the costs for quality control of drinking water, (e) the condition of the piping system and the necessary costs for repair, operation, and maintenance, and (f) the additional costs for ensuring water availability (Althoff 2007).

12.7.3 Wastewater Disposal

The German wastewater charge has been relatively stable since 2000. Between 2000 and 2009, the wastewater charge index increased by 14 percent (see Fig. 12.11). The rate increase was below the inflation rate of 15.9 percent. In 2005, using the *freshwater standard*, the consumer paid an average of 2.28 euros per cubic meter of wastewater, while under the *split charge standard* the consumer paid an average wastewater charge of 2.05 euros per cubic meter and a precipitation water charge of 0.88 euros per square meter of paved and drained plot area. On average, in 2005, the charge for collection and treatment of wastewater and precipitation water was about 10.75 euros per capita per month. Wastewater charges are regionally very different due to varying conditions such as infrastructure, water consumption, differences in local topography, demand of rehabilitation, population density, different basis of calculation, considering the Local Tax Laws of the federal

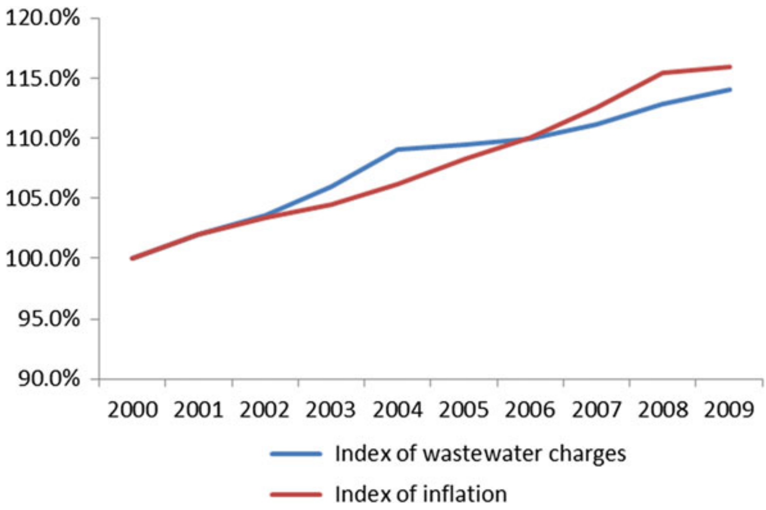

Fig. 12.11 Wastewater charge index compared to inflation from 2000 to 2009 (Reproduced from ATT et al. 2011)

states, and so on (Federal Association of the German Gas and Water Industries & German Association for Water, Wastewater, and Waste (BGW and DWA) 2005, as cited in Althoff 2007).

12.7.4 International Price Comparison

In Germany, each water connection has a water meter to measure water consumption, but that is not the case in all European countries. For example, recently in England, only 20 percent of households had water meters, while the remaining water consumption was based on estimates,' but that is now changing. By 2015, up to 92 percent of consumers in the southeast of England who get their water from "Southern Water" will be metered in line with the rest of Europe (see Chap. 5). In 2003, the drinking water costs in Germany were 82 euros per capita per year, which was ranked as the third highest costs, after England and Wales with 92 euros and France with 85 euros. The wastewater charges were 111 euros per capita per year, followed by England/Wales with 93 euros and France with 90 euros. However, factors such as state subsidies, allowances and differences in performance level are not considered in this comparison. The wastewater charges in England and Wales could be higher than those in Germany if such factors are included in the cost comparison (VEWA-Studie 2006, as cited in Althoff 2007). As noted by Althoff, "one should judge an international comparison of water price very critically, because in many cases the water prices do not reflect the real or actual costs for water supply." In Europe, as elsewhere, water prices are often influenced by political considerations as subsidies and allowances are not uncommon (German Federal Association of the German Gas and Water Industries (BGW) 2007, as cited in Althoff 2007).

On behalf of the German Association of German Gas and Water Management, the Metropolitan Consulting Group has undertaken a study to compare costs in water supply and wastewater disposal in European countries. The study is based on three levels, including (a) comparison of average turnover tax included in the country-specific prices for the consumer; (b) cost recovering water prices, including subsidies and allowances after taxes; (c) price at a uniform performance level, i.e. how high the cost level for water and wastewater would be at a performance and quality level similar to that in Germany. The result of the study indicated that "if England/Wales and France reached the German quality standard, for both countries the annual costs for drinking water would amount to 106 euros per capita and hence the costs would be higher than in Germany (84 euros per capita)" (VEWA-Studie 2006, as cited in Althoff 2007). For wastewater disposal the price in England/Wales would be 138 euros per capita, and for France the price would be 122 euros per capita. Hence both countries would be more expensive than Germany where the comparable price was 119 euros per capita. (VEWA-Studie 2006, as cited in Althoff 2007).

12.8 Benchmarking in Water Management

The development of water industry benchmarking is a joint effort between the water industry and its political partners. Since 1950, a systematic comparison has been implemented in the German water supply and wastewater disposal sector. In 2003, the water industry cooperatively developed and refined a conceptual framework for benchmarking; and later a large-scale propagation of voluntary benchmarking was undertaken (Althoff 2007). In 2005, the German Association for Water, Wastewater, and Waste signed the extended "Statement of the associations of the water industry on benchmarking in the water sector", in which the associations agreed to a regular submission of their information for comparison (Althoff 2007).

The voluntary benchmarking is carried out by independent private providers, which ensures a high quality standard of benchmarking. According to a 2011 Profile of the German Water Sector, "the benchmarking methods are continuously refined by research institutes in cooperation with practitioners from the water industry. An example of this refinement is the development of detailed performance indicators for the processes of water abstraction, processing and distribution." As a result, the German water industry is able to ensure that supply security, supply quality, economic efficiency, and sustainability are continually assessed through a benchmarking process.

12.9 Regulatory Requirements: Comparing Ontario and Germany

Like most developed countries, Germany has set Maximum Concentration Levels (MCLs) for drinking water (see Tables 12.4 and 12.5). These cover chemical and other indicator parameters, while MCLs on water quality in Ontario are divided into three categories: microbiological, chemical, and radiological parameters. All MCLs for microbiological parameters are zero in Germany and Ontario. That is, Escherichia coli (*E. Coli*) and total coliforms should not be detectable in drinking water samples. Although Germany does not set MCLs for radiological parameters, in comparison with Ontario a number of MCLs for chemical parameters are considerably lower, including 1,2–dichloroethane, antimony, boron, cadmium, nitrite, tetrachloroethene, trichloroethene, trihalomethanes, uranium, and vinyl chloride. The only German MCL for a chemical parameter that is substantially higher than that of Ontario is for nitrate (50 mg/l), when compared to the Ontario MCL of 10 mg/l, while as shown above, the wastewater treatment achieves a significant reduction of nitrate in Germany and since 2010, the nitrate concentrations comply with the limit value of 50 mg/L. Some chemicals such as N-nitrosodimethylamine (NDMA), polychlorinated biphenyls (PCB), dichlorodiphenyltrichloroethane (DDT) + metabolites, and pentachlorophenol have no required MCLs in Germany, but are regulated in Ontario, while acrylamide, copper, epichlorohydrin, nickel, and polycyclic aromatic hydrocarbons have stated MCLs in Germany, but are not regulated in Ontario. We expect that NDMA and PCBs are not regulated in Germany because the wastewater treatment is of a sufficiently high standard that these contaminants are removed at the wastewater treatment stage. However, unlike Ontario, there are unregulated but Health-Related Indicator Values for all micropollutants in Germany. This is a significant advancement in the quest for contaminant free drinking water.

12.10 Conclusion

Our review of the available literature shows that the German water sector is well organized. Although the prices are high, when taking quality and performance into account, the price is in fact lower than some other European countries. Furthermore, German authorities regard the conservation of water as a social objective and so they put a high price on water in order to promote conservation. This is credible as Germany has the lowest level of leaks, as shown above. Another important social objective is the maintenance and restoration of ecosystem functioning, a very laudable objective.

By volume, more of the water is supplied by private sector companies, although there are more public sector utilities. Some 74 percent of drinking water comes from groundwater sources, which is typically of high quality before it is treated in the drinking water treatment plants. It is likely to be free of many micropollutants.

Only 0.03 percent of wastewater is not treated by a biological wastewater treatment process. About 26 percent of drinking water is from surface water, namely rivers and lakes that receive biologically treated wastewater. We can conclude from the evidence presented in this chapter that German drinking water quality is better than Class 5, a classification of treated water developed in Chap. 3. Class 5 treatment removes all other pollutants including micropollutants (PPCPs) discussed in this chapter.

There is evidence of very low water leakage, due to a well-maintained water infrastructure, the costs of which are financed largely by water charges and fees. Hence there is no “deferred maintenance”, a characteristic of many North American cities. Based on the evidence, we are able to conclude that German consumers enjoy very high quality drinking water.

Appendix

Table 12.4 German drinking water maximum concentration level for chemical parameters (Bundesgesetzblatt 2011)

Chemical parameters			
Parameter	MCL (mg/l)	Parameter	MCL (mg/l)
Acrylamide	0.0001	Antimony	0,005
Benzene	0.001	Arsenic	0.01
Boron	1	Benzo [a] pyrene	0.00001
Bromate	0.01	Lead	0.01
Chrome	0.05	Cadmium	0.003
Cyanide	0.05	Epichlorohydrin	0.0001
1,2-dichloroethane	0.003	Copper	2
Fluoride	1.5	Nickel	0.02
Nitrate	50	Nitrite	0.5
Plant protection products and biocidal products	0.0001	Polycyclic aromatic hydrocarbons	0.0001
Plant protection products and biocidal total	0.0005	Trihalomethanes	0.05
Mercury	0.001	Vinyl chloride	0.0005
Selenium	0.01	Uranium	0.01
Tetrachloroethene	0.01		
Trichloroethene	0.01		

Note Measured quantities are based on a representative for the weekly average value ingested by consumers; this is provided in Article 7, Paragraph 4 of the Drinking Water Directive, which calls for the establishment of a harmonized procedure. The competent authorities are required to ensure that all appropriate measures are taken to reduce the concentration of lead in water intended for human consumption to achieve the limit as far as possible. Actions designed to achieve this are progressively given priority where the lead concentration in water for human consumption is higher than the MCL of 0.01 mg/L

Table 12.5 German drinking water maximum concentration level for indicator parameters (Bundesgesetzblatt 2011)

Indicator parameters		
Parameter	Unit	MCL(mg/l)
Aluminum	mg/L	0.2
Ammonium	mg/L	0.5
Chloride	mg/L	250
Clostridium perfringens (including spores)	Number/100 ml	0
Coliform bacteria	Number/100 ml	0
Iron	mg/L	0.2
Staining (Spectral absorption coefficient at 436 nm)	1/m	0.5
Odor	ton	3 at 23 °C
Taste		Acceptable to consumers and no abnormal change
Colony count at 22 °C		No abnormal change
Colony count at 36 °C		No abnormal change
Electrical conductivity	µS/cm at 25 °C	2790
Manganese	mg/L	0.05
Sodium	mg/L	200
Organic carbon	mg/L	No abnormal change
Oxidizability	mg/L O_2 demand	5
Sulfate	mg/L	250
Cloudiness	Nephelometric turbidity units (NTU)	1
Hydrogen ion concentration	pH units	6.5 to 9.5
Calcite	mg/L $CaCO_3$	5
Tritium	Bq/L	100
Total indicative dose	mSv/year	0.1

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