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Understanding the influence of climate change on the embodied energy of water supply

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ABSTRACT

The current study aims to advance understandings on how and to what degree climate change will affect the life cycle chemical and energy uses of drinking water supply. A dynamic life cycle assessment was performed to quantify historical monthly operational embodied energy of a selected water supply system located in northeast US. Comprehensive multivariate and regression analyses were then performed to understand the statistical correlation among monthly life cycle energy consumptions, three water quality indicators (UV_{254}, pH, and water temperature), and five climate indicators (monthly mean temperature, monthly mean maximum/minimum temperatures, total precipitation, and total snow fall). Thirdly, a calculation was performed to understand how volumetric and total life cycle energy consumptions will change under two selected IPCC emission scenarios (A2 and B1). It was found that volumetric life cycle energy consumptions are highest in winter months mainly due to the higher uses of natural gas in the case study system, but total monthly life cycle energy consumptions peak in both July and January because of the increasing water demand in summer months. Most of the variations in chemical and energy uses can be interpreted by water quality and climate variations except for the use of soda ash. It was also found that climate change might lead to an average decrease of 3–6% in the volumetric energy use of the case study system by the end of the century. This result combined with conclusions reached by previous climate versus water supply studies indicates that effects of climate change on drinking water supply might be highly dependent on the geographical location and treatment process of individual water supply systems.

1. Introduction

Over the last few decades, water shortage has led to increased adoption of alternative water sources such as imported water, desalinated seawater, and even reclaimed water in many densely populated areas around the world (Lazarova et al., 2012; Yüce et al., 2012; Bischel et al., 2011; Elimelech and Phillip, 2011; Martinez and Clark, 2012; Jiang et al., 2013). While these alternative water sources serve as an important supplement of the dwindling freshwater supply, their adoptions are usually associated with significant short-term and long-term costs in forms of life cycle energy, economic costs, and environmental impacts (Stokes and Horvath, 2009; Mo et al., 2014). For example, producing 1 m^3 of tap water through fresh ground or surface water sources typically consumes around 0.5 kWh of electricity (Goldstein and Smith, 2002), whereas water importation in California requires around 1.6–2.6 kWh and seawater desalination via reverse osmosis uses around 4.4–5.5 kWh to produce the same amount of water (Mo et al., 2014; Klein et al., 2005). These energy and environmental burdens could potentially lead to new or elevated stresses in energy supply, public funds, as well as ecosystem services, which may eventually be partially or fully reflected in water and energy prices, causing ripple effects on social equity and economic development (Kaika, 2003; Rogers et al., 2002). One example is the South-to-North Water Diversion Project in China which is likely to more than double water prices in receiving cities due to the vast project constructional costs and pumping demands (a power capacity of 454 MW to pump water from Yangtze River through the Eastern Route) (Berkoff, 2003; Kuo, 2014).

Climate change is likely to further increase the energy and cost of water supply through combined effects on water quality and availability, service infrastructure, and user demands, challenging

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the sustainable management of both water and energy resources. In the US, the hydrologic cycle is accelerating with increasing flooding and downpours in the northeast as well as more frequent droughts and shrinking snowpack storage in the southwest (Stocker et al., 2013; Barnett et al., 2005). Sea level rise and subsequent seawater intrusion have threatened freshwater availability and quality in many coastal regions (Stocker et al., 2013; Shannon et al., 2008). Already water-stressed regions such as California, Texas, and Arizona are particularly vulnerable to climate change because they are predicted to have the highest temperature increase as well as the greatest precipitation reduction (Stocker et al., 2013; Milly et al., 2005; Bates et al., 2008). Collectively, these climate change effects could impose escalated challenges in providing reliable and low cost water services in the foreseeable future. Utility managers and city planners need to be prepared for such changes so that the most appropriate mitigation and adaptation strategies can be implemented. Therefore, a systematic understanding on the influence of climate change on water supply services especially its indirect effect on energy utilities is imperative, given the lead-time needed for decision making, planning, and construction in water and energy utilities and government agencies.

The energy use of varied forms of water supply has been investigated via different approaches. Traditional energy audits and risk assessments quantify the direct energy used onsite of water systems (Wilkinson, 2000; Elliott et al., 2003; Means, 2004). In the past decade, a proliferated number of life cycle assessments (LCAs) was conducted to examine the life cycle energy use of water supply systems (Stokes and Horvath, 2009; Mo et al., 2014, 2010, 2011; Rothausen and Conway, 2011; Racoviceanu et al., 2007; Lassaux et al., 2007; Godskesen et al., 2010; Lundie et al., 2004; Friedrich et al., 2009; Landu and Brent, 2007; Lyons et al., 2009). These studies have revealed the importance of indirect energy flows associated with providing chemicals and services in water systems in addition to direct energy consumptions, and provided a more comprehensive approach in quantifying the "true" energy embodiment in water systems to inform sustainable management and decision making. Nevertheless, most of these LCAs are static studies focusing on evaluating the energy uses of water supply at given times ("snapshots"), while critical information regarding the trends and dynamic patterns of the life cycle energy in response to exogenous drivers such as climate change is missing (Stokes and Horvath, 2009; Mo et al., 2010, 2011; Racoviceanu et al., 2007; Lyons et al., 2009; Friedrich, 2002). These "snapshots" are mostly taken for a current or past time. Only a few studies have investigated future energy uses based on projected water demand and freshwater availabilities; however, climate change was not considered in either water demand or water availability estimations (Mo et al., 2014; Lundie et al., 2004).

The impacts of climate change on water availability have been widely investigated (Milly et al., 2005; Yates, 1996; Gleick, 1987; O'Hara and Georgakakos, 2008; Muttiah and Wurbs, 2002; Bekele and Knapp, 2010; Matonse et al., 2013), while only a few studies have offered discussion on the influence of climate change on water quality (Whitehead et al., 2009; Mimikou et al., 2000; Senhorst and Zwolsman, 2005; Zwolsman and Van Bokhoven, 2007; Arheimer et al., 2005; Delpla et al., 2009). Nonetheless, water quality could have more acute effects on treatment energy and cost compared with water availability. For instance, increased water temperature and summer drought can lead to enhanced growth of algae and cyanobacteria, cascading the formation of disinfection byproducts and treatment costs (Mimikou et al., 2000; Zwolsman and Van Bokhoven, 2007; Delpla et al., 2009). Storm events and flooding could result in elevated suspended solids, nutrients, and pollutants (e.g., pesticides) fluxes (Whitehead et al., 2009). Seawater intrusion increases groundwater salinity and its associated treatment difficulty in many coastal regions (Barlow and Reichard, 2010). On the other hand, warmer water may increase the reaction rates of treatment processes as well as physical operation of facilities, which may potentially improve treatment efficiency and reduce cost (Crittenden et al., 2012). The degree of these positive and negative effects could vary considerably across regions based on local baseline water and climate profiles, water treatment technologies, and socioeconomic conditions, yet little is known about such tradeoffs to guide management practices.

Therefore, this study primarily investigates the potential effects of climate change on water treatment through changes in water quality parameters for a case study water supply system located in the northeastern US. To achieve this goal, an assessment framework including dynamic life cycle energy assessment, multivariate analysis, and regression analysis were adopted. This study aims to assist proactive management of water and energy resources with the ultimate goal of improving their long term resiliency and sustainability under global changes.

2. Methodology

2.1. Water quality indicators

Raw water quality is a key factor in determining the selection, design, and operation of water treatment processes (Crittenden et al., 2012). Table 1 provides a list of water quality indicators as well as their influences on six individual treatment processes, including coagulation, filtration, membrane separation, disinfection, ion exchange, and air stripping and aeration. It has to be noted that Table 1 does not exhaust all water quality parameters that are potentially significant to human and ecological health (e.g., heavy metals, nutrients, dissolved oxygen etc.); however, the listed indicators are closely related to chemical dosages, equipment utilization rates, and pre- and post-treatment requirements in drinking water systems' design and actual operation. Most of these water quality indicators are likely to be influenced by climate change (Delpla et al., 2009), which could further affect the daily operation of existing treatment plants as well as their energy demands.

2.2. Study site description

The case study water supply system (CSS) is located on the coast of northeast US serving a population of around 2.55 million. Raw water of the CSS comes from two protected inland reservoirs, which are filled naturally by rain and snow fall on the surrounding watersheds. The two reservoirs have high altitudes, and hence the influence of sea level rise on the water quality is minimum. The CSS utilizes ozone (generated from liquid oxygen) as the primary disinfectant and chloramine (formed by sodium hypochlorite and aqueous ammonia) for residual disinfection. Additionally, sodium bisulfite is used for ozone removal, and sodium hydrofluorosilicic acid is used for tooth health protection. Towards the end of the treatment process, sodium carbonate (soda ash) and carbon dioxide are used for alkalinity and pH adjustment respectively. Three types of energy are directly used onsite of the CSS: 1) electricity is used for pumping, mixing, facility administration etc.; 2) natural gas is primarily used for space and water heating; and, 3) diesel is used as backup power supply. In particular, the local electricity provider has been paying the CSS to go off grid during storms and other extreme climate events in order to relieve regional energy stress and to reduce outages. This interaction between the CSS and the electricity provider further implies the importance of understanding the climate-water-energy nexus and finding solutions to reduce the energy use in water systems.

Monthly flow rates as well as the chemical and energy uses over