Cost comparison of seawater for toilet flushing and wastewater recycling

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Abstract

As freshwater resources are becoming increasingly scarce, unconventional sources of water should be given new consideration. In coastal cities, seawater, with minimal treatment, can be used for toilet flushing, reducing the demand for freshwater. Currently, it is practised on a large scale only in Hong Kong. This study estimates the cost of seawater flushing and compares it to the cost of wastewater recycling for 15 major coastal cities around the world: Buenos Aires, Chennai, Hong Kong, Jakarta, Karachi, Los Angeles, Miami, Mumbai, New York City, Osaka, San Francisco, Shanghai, Singapore, Sydney and Tokyo. While seawater flushing requires a separate network of mains and, therefore, a greater capital cost, wastewater recycling has a higher ongoing treatment cost. Wastewater recycling, depending on the potability of the recycled water, may also require a separate network of mains, but one with a lower maintenance cost due to its lower vulnerability to corrosion compared to seawater mains. This study finds Chennai, Mumbai and Shanghai to have strong potentials for seawater flushing. That these cities have among the highest population densities in the world and are in the developing world explains their relatively lower unit costs for seawater mains.

Keywords: Integrated water management; Seawater; Toilet flushing; Urban water management; Wastewater recycling; Water reuse; Water scarcity; Water supply

1. Introduction

Freshwater resources are becoming increasingly scarce (Postel, 2000) due to the overdrawing of ground and surface waters that have led to depleted water tables and stream flows, and population growth that has increased the competition for water. Further, according to the Intergovernmental Panel on Climate Change (IPCC, 2007), climate change is expected to lead to more frequent, longer lasting extreme low flow events. Pollution has also adversely affected source waters, rendering them unsuitable for drinking and other uses. In many places, there is an urgent need to identify and develop alternative water resources.

doi: 10.2166/wp.2014.045

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One water resource available to coastal cities is seawater. Seawater desalination is already a source of drinking water in many places with membrane reverse osmosis emerging as the preferred technology for new desalination facilities (Greenlee et al., 2009). Seawater, with minimal treatment, can also be used for toilet flushing, though thus far this is widely practised only in Hong Kong. Another water resource is recycled wastewater. Wastewater recycling is more common and adopted on a large scale in Singapore (Tortajada, 2006), Southern California (Chalmers et al., 2003), Australia (Radcliffe, 2006), Israel (Icekson-Tal et al., 2003) and other parts of the world.

Compared to wastewater recycling, seawater flushing requires considerably less treatment, and thus, consumes less energy, resulting in a lower operating cost. However, it requires the establishment of a separate network of mains and, therefore, has a higher capital cost. (Wastewater recycling to produce non-potable water requires a separate network of mains as well, but not if it is to produce potable water, whether for direct or indirect use.) Moreover, due to corrosion, the cost of maintaining that separate network of mains is greater than for a similar network of mains carrying freshwater. For places far from the coast or at high elevations, seawater flushing entails an additional cost of seawater transportation that may be prohibitive.

This study estimates and compares the unit costs of seawater flushing and wastewater recycling for 15 major coastal cities around the world: Buenos Aires, Chennai, Hong Kong, Jakarta, Karachi, Los Angeles, Miami, Mumbai, New York City, Osaka, San Francisco, Shanghai, Singapore, Sydney and Tokyo. These cities face growing water demands, which they struggle to meet sustainably and cost-effectively. Some of them are highly dependent on water transported over long distances from progressively vulnerable surface sources, while others rely on desalinated seawater produced from energy intensive processes or groundwater from fast diminishing aquifers.

Here, wastewater recycling has been selected for comparison against seawater flushing, and not seawater desalination, because wastewater recycling is usually significantly cheaper than seawater desalination (Wittholz et al., 2008) due to the higher salinity of raw seawater, as compared to treated wastewater. Therefore, for the purpose of this study, it is more conservative to compare seawater flushing against wastewater recycling than against seawater desalination. In other words, in situations where seawater flushing is cheaper than wastewater recycling, it can also be expected to be cheaper than seawater desalination.

To achieve the objective of this study, the costs of seawater flushing for the different cities are extrapolated from the known cost of constructing seawater mains in Hong Kong. The costs of wastewater recycling are estimated from cost relationships reported in the literature (Wittholz et al., 2008) derived from real costs. Results are obtained for various assumptions of energy cost, pipe material and per capita consumption of flush water. The results are useful to policymakers to understand the conditions under which seawater flushing is favourable and should be seriously considered as a new source of water, where conventional surface and ground water sources are no longer sufficient or sustainable.

2. Background information

2.1. Seawater flushing

Currently, seawater flushing is adopted on a large scale only in Hong Kong. According to the Hong Kong Water Supplies Department (WSD), seawater has been used for toilet flushing there since the
1950s, and currently covers 80% of the population, up from 65% in 1991 and 78.6% in 1999. Seawater fulfills about 20% of the city’s total water demand. In 2010, 270 million m$^3$ of seawater were consumed. This reduces the city’s need for freshwater, 70–80% of which it meets by purchasing water from nearby Guangdong in mainland China and piping it over 80 km, and the remaining 20–30% by drawing from local reservoirs.

In Hong Kong, seawater is delivered to end users via a dedicated system of 40 pumping stations, 1,500 km of mains and 45 service reservoirs. However, due to the high salinity of the water, the mains and pumping equipment are susceptible to corrosion. Based on actual statistics from the WSD, Tang et al. (2007) found the average frequency of pipe bursts for the seawater system to be three times greater than for the city’s freshwater system. Leaking seawater has also led to the degradation of concrete and metal structures and fittings. In view of these problems, polyvinyl chloride (PVC) pipes are increasingly being used in place of traditional metal pipes, though PVC pipes are generally lighter and, thus more vulnerable to breakage.

In Hong Kong, the seawater for toilet flushing receives minimal treatment before use. The water is first channeled through mesh screens to remove debris, then subjected to electro-chlorination for disinfection. (Electro-chlorination is a process where electrical currents are passed through the seawater to convert salt to sodium hypochlorite, a disinfectant.) Disinfection deactivates pathogenic microorganisms and suppresses biological growth in pipes. Additional treatment has been found unnecessary to render the seawater fit for use in terms of colour, turbidity, odour, dissolved oxygen, biological oxygen demand (BOD) and suspended solids, etc. (Tang et al., 2007).

Also in Hong Kong, the saline sewage resulting from seawater flushing is mixed with wastewater from freshwater uses, treated using conventional wastewater treatment methods, then released to the sea. Given that seawater flushing contributes approximately 20% of the total water demand, the ensuing mixture has a salinity that is about a fifth the salinity of raw seawater. According to Tang et al. (2007), based on literature findings and Hong Kong’s operational experience, this level of salinity does not significantly affect BOD removal and denitrification. It does, however, reduce nitrification, but this can be somewhat mitigated with adjustments to the activated sludge process.

Saline sewage opens new possibilities for innovations in wastewater treatment. Consider the sulfate reduction, autotrophic denitrification and nitrification integrated (SANI) process (Wang et al., 2009). SANI makes use of the relatively high concentration of sulfate in saline sewage to induce sulfate reduction which, in turn, produces hydrogen sulfide to provide electron donors for autotrophic denitrification. Both these processes have low growth yields. SANI combines them with nitrification, another low growth yield process, to achieve up to 95 and 74% removals of chemical oxygen demand and nitrogen, respectively, without the high level of sludge production commonly associated with conventional wastewater treatment. This reduces sludge disposal costs, which can be significant especially in places like Hong Kong where existing landfills are reaching their maximum capacities and where no more land is available for new landfills.

2.2. Wastewater recycling

Wastewater recycling refers to the treatment and reuse of wastewater treatment plant effluent that would otherwise be discharged and lost to receiving waters. It conserves the water within the system, reducing the demand for new supplies. It also improves supply reliability, as the availability of recycled wastewater is relatively stable and unaffected by weather fluctuations. Wastewater recycling is now
practised in many places to supplement conventional sources of water (Anderson, 2003; Miller, 2006; Angelakis & Durham, 2008). However, in most places, it is still not fully accepted as a direct source of potable water by the general public (Friedler & Lahav, 2006; Hurlimann & Dolnicar, 2010), partly due to health risks from microbial pathogens and chemical contaminants resistant to treatment, whose survival are yet to be entirely understood (Toze, 2006).

Singapore’s NEWater project is a well-known example of wastewater recycling on a large scale (Tortajada, 2006). According to the Singapore Public Utilities Board, the first NEWater plant started operations in 2003. As of the end of 2010, there were five NEWater plants, the newest and largest having the capacity of 70 million m³/year. The plants apply microfiltration, reverse osmosis and ultraviolet (UV) disinfection to produce water of a very high purity from wastewater effluent. Much of the treated water is sold to the semiconductor industry, and the remaining water is returned to water supply reservoirs for indirect potable use. Currently, NEWater supplies 30% of the nation’s total water demand. By 2060, that percentage is expected to increase to 50%.

Another example of wastewater recycling is the Groundwater Replenishment System (GWRS) (Chalmers et al., 2003) in Southern California, which began operation in 2008. Southern California is a semi-arid region that is heavily reliant on groundwater and the long-distance transport of surface water to satisfy its needs. Like the NEWater plants, the GWRS uses microfiltration, reverse osmosis and UV disinfection to treat wastewater effluent. In addition, it mixes the water with hydrogen peroxide, a disinfectant and also an oxidant, to destroy trace organic compounds. The system produces 90 million m³/year of recycled water used to artificially recharge the Orange County groundwater basin, to prevent seawater intrusion in the aquifer and to store the water for future potable and non-potable uses.

An Australian example of wastewater recycling is the Rouse Hill Wastewater Recycling Scheme (RHWRS) in Sydney. According to Sydney Water, the scheme started in 2001 and presently generates 1.7 million m³/year of recycled water. Like the NEWater project and GWRS, it applies microfiltration and UV disinfection to wastewater effluent, but uses nanofiltration in place of reverse osmosis. Nanofiltration has a lower operating cost, but produces water of lesser quality (Bellona & Drewes, 2007). Under the RHWRS, the recycled water is piped to homes and businesses for outdoor use and toilet flushing, and is kept separate from potable water by means of a dedicated network of mains.

3. Methods

The unit costs of seawater for toilet flushing and wastewater recycling are estimated for 15 major coastal cities around the world. For wastewater recycling, two scenarios are considered. Under the first scenario, Scenario A, the recycled water is assumed safe for mixing with drinking water supplies, and a separate network of mains is unnecessary. Under the second scenario, Scenario B, the recycled water is for non-potable use, and a separate network of mains is necessary to prevent mixing with potable supplies.

3.1. Seawater for toilet flushing

A large portion of the overall cost of seawater for toilet flushing is the capital cost of constructing a dedicated network of mains to distribute the seawater. This cost may be estimated by Equation (1)
(Swamee & Sharma, 2008)

\[ CSP_i = k_{CSP} L_i D_i^m \]  

\( CSP_i \) is the capital cost of seawater mains in city \( i \), \( k_{CSP} \) a proportionality constant, \( L_i \) the total length of the mains in the city, \( D_i \) the average diameter of the mains and \( m \) an exponent. For ductile iron cement lined pipes, commonly used for water distribution, \( m \) can be assumed as 0.935 (Samra & Essery, 2003). For Hong Kong, \( CSP_i \) is approximately US$0.507 billion in 2000 dollars (Tang et al., 2007) or US$0.519 billion in 2005 dollars, assuming a cost index of 418.8 in 2000 and 428.5 in 2005, based on the city’s Civil Engineering Works Index. For a 20-year lifetime and an interest rate of 4%, this is equivalent, in 2005 dollars, to an annuity of US$0.038 billion/year. (The lifetime of 20 years is the same as assumed by Tang et al. (2007) and is shorter than the usual lifetimes of water infrastructures, to account for the effects of enhanced corrosion from carrying saline water.) Since the cost for Hong Kong, represented by the subscript HKG, is known, the costs for other cities can be extrapolated using Equation (2) below, assuming a constant \( k_{CSP} \)

\[ \frac{CSP_i}{CSP_{HKG}} = \left( \frac{L_i}{L_{HKG}} \right) \left( \frac{D_i}{D_{HKG}} \right)^{0.935} \]  

(2)

For a city \( i \), \( L_i \) is a function of the area covered by seawater flushing, \( A_i \). The relationship between \( L_i \) and \( A_i \) may be approximated by Equation (3) (Tang et al., 2006)

\[ \left( \frac{L_i}{L_{HKG}} \right) = \left( \frac{A_i}{A_{HKG}} \right)^{0.5} \]  

(3)

\( D_i \) for city \( i \) depends on its magnitude of seawater supply, \( Q_i \) (Equation (4)). Generally

\[ \left( \frac{Q_i}{Q_{HKG}} \right) = \left( \frac{v_i}{v_{HKG}} \right) \left( \frac{D_i}{D_{HKG}} \right)^2 \]  

(4)

\( v_i \) is the average velocity of the seawater in the mains. Assuming a more or less constant \( v_i \) from city to city, Equation (4) simplifies to Equation (5)

\[ \left( \frac{q_i P_i}{q_{HKG} P_{HKG}} \right) = \left( \frac{D_i}{D_{HKG}} \right)^2 \Rightarrow \left( \frac{D_i}{D_{HKG}} \right) = \left( \frac{q_i P_i}{q_{HKG} P_{HKG}} \right)^{0.5} \]  

(5)

where \( P_i \) is the population of city \( i \) covered by seawater flushing, and \( q_i \) is the city’s per capita rate of consumption of flush water. Refer to Table 1 for the values of \( P_i \) assumed and the corresponding values of \( A_i \). These values are taken from various sources including the US Census Bureau, Office of Shanghai Chronicles, Hong Kong Census and Statistics Department, Singapore Department of Statistics, City of Sydney, Census Organization of India, City of Osaka, Tokyo Metropolitan Government, National...
Substituting Equations (3) and (5) into Equation (2) gives Equation (6)

\[ \frac{CSP_i}{CSP_{HKG}} = \left( \frac{A_i}{A_{HKG}} \right)^{0.5} \left( \frac{q_i P_i}{q_{HKG} P_{HKG}} \right)^{0.4675} \]  \tag{6}

\( CSP_i \) as estimated by Equation (6) is the cost of seawater mains in city \( i \) given the costs of labour, materials and overheads in Hong Kong. To account for differences in these costs between countries, \( CSP_i \) is multiplied by the ratio of the construction purchasing power parity (CPPP) of city \( i \) to that of Hong Kong. The CPPPs for 2005 for the countries containing the 15 cities examined in this study are available from the World Bank’s International Comparison Program (http://data.worldbank.org/data-catalog/international-comparison-programwww.worldbank.org/data/icp). As of the time of writing, in July 2013, CPPPs were only available for 2005 and, therefore, \( CSP_i \) is based on a \( CSP_{HKG} \) that is in 2005 dollars. To adjust \( CSP_i \) to consider locational effects and to inflate it to 2010 dollars (Equation (7))

\[ ACSP_{i,2010} = CSP_{i,2005} \left( \frac{PPP_{i,2005}}{PPP_{HKG,2005}} \right) \left( \frac{CI_{i,2010}}{CI_{i,2005}} \right) \]  \tag{7}

\( ACSP_{i,2010} \) is the final estimate of the cost of seawater mains in city \( i \) in 2010 dollars that has been adjusted for location and time. \( CSP_{i,2005} \) is \( CSP_i \) in 2005 dollars, and \( PPP_{i,2005} \) the CPPP for

\[ aCPPP: \text{construction purchasing power parity}; \text{LCU: is local currency unit.} \]
city \( i \) in 2005. \( CI_{i,2010} \) is an appropriate cost index for \( i \) in 2010 and \( CI_{i,2005} \) the same index in 2005.

Table 1 shows the construction purchasing power parities and values of cost indices assumed. For the cost indices, construction related indices are used where available and accessible, and where otherwise, consumer price indices are used. Construction related cost indices are available from a variety of sources, including the Hong Kong Civil Engineering and Development Department, Singapore Building and Construction Authority, Australian Bureau of Statistics, Construction Industry Development Council of India, Statistics Bureau of Japan and Engineering News-Record. Consumer price indices are available from the World Bank.

For a city \( i \), \( ACSP_{i,2010} \) as estimated using Equation (7) assumes traditional metal mains since, historically, the seawater mains in Hong Kong consist mostly of metal pipes (Tang et al., 2007). To estimate the cost of PVC mains, they are assumed as 85% that of traditional metal mains, based on Spencer (2008) who reported the cost of PVC pipes to be on average about 85% that of cast iron pipes. It is also assumed that PVC mains have a lifetime of 60 years instead of the 20 years assumed for metal mains.

Here, any additional cost of wastewater treatment arising from the increase in salinity from seawater flushing is assumed insignificant. This is based on the operational experience of Hong Kong, where saline sewage from toilet flushing is mixed with freshwater sewage from other uses and treated using conventional wastewater treatment processes. According to Tang et al. (2007), the added salinity from seawater flushing has been found to have no significant effects on these treatment processes.

3.2. Wastewater recycling

3.2.1. Scenario A. Under this scenario, wastewater effluent is treated to standards considered acceptable for potable use, and therefore a separate network of mains to transport the recycled water is unnecessary. However, to treat the water to this level, more advanced technology is utilized that requires a higher consumption of energy, incurring greater costs. This scenario is modeled after the NEWater project in Singapore and the GWRS in Southern California. Equation (8) is used to estimate the capital cost of such a system \( i \).

\[
\frac{CWR_i}{CWR_0} = \left( \frac{CAPA_i}{CAPA_0} \right)^n
\]  

(8)

\( CWR_i \) is the capital cost of the system and \( CWR_0 \) of a reference system. \( CAPA_i \) is the capacity of \( i \) and \( CAPA_0 \) of the reference system. \( n \) is an exponent. From Wittholz et al. (2008), who surveyed the actual costs of over 300 projects, assuming the wastewater recycling is by reverse osmosis, \( CWR_0 \), \( CAPA_0 \) and \( n \) are taken as US$93.5 million in 2006 dollars, 275,000 m³/day and 0.74, respectively.

Also, from Wittholz et al. (2008), the breakdown between capital and operating costs is assumed as 35% capital and 65% operating, and of the operating cost, typically, about 46% is energy cost. This gives Equations (9)–(11):

\[
UWR_i = \frac{1}{0.35 \times 0.9 \times CAPA_i} CWR_i (A/P, 0.04, 20)
\]  

(9)
\[
UWRO_i = 0.35\ UWR_i \\
UWRE_i = 0.30\ UWR_i
\]

\(UWR_i\) is the unit cost of system \(i\) inclusive of both capital and operating costs, and \(UWRO_i\) and \(UWRE_i\) the non-energy operating cost and energy cost of the system, respectively. \((A/P, 0.04, 20)\) is the capital recovery factor given an interest rate of 4% and infrastructure lifetime of 20 years. The value of 0.9 in the denominator of the right hand side of Equation (9) represents plant availability, which is the fraction of time the system is expected to be in operation. Both assumptions of a 20-year lifetime and plant availability of 0.9 are consistent with Wittholz \textit{et al.} (2008).

The parameters provided by Wittholz \textit{et al.} (2008) are for costs in 2006 dollars; however, the final estimates of costs of seawater flushing in the previous subsection are in 2010 dollars. Though indices exist to account for changes in costs with time, they generally do not apply to reverse osmosis equipment, the costs of which, contrary to the effects of these indices, have shown a downtrend over past years due to technological advancement (Dore, 2005). Thus, it is assumed any upward impact of inflation on the cost of wastewater recycling is cancelled out by any downward impact of technological advancement and, therefore, any net change in total cost from 2006 to 2010 is negligible.

It is also assumed that the effects of location on the cost of wastewater recycling are negligible. While individual cost components may differ from country to country, as demonstrated by Park \textit{et al.} (1997), these differences tend to balance out such that, overall, total cost does not change significantly with location.

### 3.2.2. Scenario B.

Under this scenario, wastewater effluent is treated to standards that, although they are high, are still not considered high enough to allow the treated water to be mixed with potable water. Therefore, under this scenario, to distribute the recycled water to end users for non-potable uses, a separate network of mains needs to be constructed. Another difference between this scenario and Scenario A is the use of nanofiltration in place of reverse osmosis. Nanofilters operate at lower pressures, but have larger pore sizes that prevent them from capturing very fine particles. Scenario B is modeled after the RHWRS in Sydney, Australia.

To estimate the capital cost of constructing a separate network of mains, the same procedures outlined in Section 3.1 are followed. The only difference between the cost of mains for seawater and that for recycled water is the lifetime of the mains. For the former, the lifetime is assumed as 20 years for metal mains and 60 years for PVC mains. For the latter, a lifetime of 60 years is assumed for both metal and PVC mains.

The capital cost of constructing a wastewater recycling facility under Scenario B is assumed to be the same as that for Scenario A. Even though nanofiltration membranes can be expected to be cheaper than reverse osmosis membranes, they constitute just a portion of the overall cost, which also includes the costs of design, construction and other equipment, and therefore, whatever differences there are they can be assumed to be negligible. For the same reason, the non-energy operating cost of the facility is assumed to be the same as under Scenario A.

However, the energy cost of the facility can be expected to be lower under the current scenario. According to Bellona & Drewes (2007), who made a side-by-side comparison between nanofiltration and reverse osmosis, nanofiltration utilizes two to four times less energy than reverse osmosis. Therefore, the energy cost of a wastewater recycling plant under the current scenario may be approximated as about one-third of that under Scenario A.
4. Results and discussion

Table 2 gives estimates of the unit costs of seawater for toilet flushing and wastewater recycling for the coastal cities of Buenos Aires, Chennai, Hong Kong, Jakarta, Karachi, Los Angeles, Miami, Mumbai, New York City, Osaka, San Francisco, Shanghai, Singapore, Sydney and Tokyo. Figure 1 plots the unit cost of seawater mains, which constitutes the bulk of the cost of seawater flushing, against population density. For comparison, Figure 1 also plots the unit cost of Scenario A of wastewater recycling. The results shown are for the case of traditional metal mains for distributing both seawater and recycled wastewater (under Scenario B of wastewater recycling). Also, for the two scenarios of wastewater recycling, the results assume a recycling plant capacity of 300,000 m³/day, which is about the size of the GWRS in California and a reasonable magnitude to assume for the large-scale implementation of wastewater recycling.

From the results, it can be seen that there is a strong nonlinear relationship between population density and cost of seawater mains. Generally, the higher the former, the smaller the latter. Thus, it is not surprising to find Chennai, Mumbai and Shanghai as having the cheapest costs of seawater mains. These cities have among the highest population densities in the world. That these cities are situated in the developing world where costs of labour, materials and energy are relatively cheaper, as portrayed by their respective purchasing power parities, also explains their relatively lower costs of seawater mains. Conversely, the city where seawater mains are the most expensive is Los Angeles, followed

Table 2. Annual equivalent costs of seawater for toilet flushing and Scenarios A and B of wastewater recycling in US$/m³ in 2010 dollars, assuming traditional metal mains.

<table>
<thead>
<tr>
<th>City</th>
<th>Seawater Flushing</th>
<th>Wastewater Recycling – A</th>
<th>Wastewater Recycling – B</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Cost of mains</td>
<td>Total cost</td>
<td>Cost of treatment¹</td>
</tr>
<tr>
<td>Buenos Aires</td>
<td>0.154</td>
<td>0.154</td>
<td>0.167</td>
</tr>
<tr>
<td>Chennai</td>
<td>0.075</td>
<td>0.075</td>
<td>0.167</td>
</tr>
<tr>
<td>Hong Kong</td>
<td>0.159</td>
<td>0.159</td>
<td>0.167</td>
</tr>
<tr>
<td>Jakarta</td>
<td>0.139</td>
<td>0.139</td>
<td>0.167</td>
</tr>
<tr>
<td>Karachi</td>
<td>0.144</td>
<td>0.144</td>
<td>0.167</td>
</tr>
<tr>
<td>Los Angeles</td>
<td>0.864</td>
<td>0.864</td>
<td>0.167</td>
</tr>
<tr>
<td>Miami</td>
<td>0.785</td>
<td>0.785</td>
<td>0.167</td>
</tr>
<tr>
<td>Mumbai</td>
<td>0.083</td>
<td>0.083</td>
<td>0.167</td>
</tr>
<tr>
<td>New York City</td>
<td>0.461</td>
<td>0.461</td>
<td>0.167</td>
</tr>
<tr>
<td>Osaka</td>
<td>0.561</td>
<td>0.561</td>
<td>0.167</td>
</tr>
<tr>
<td>San Francisco</td>
<td>0.621</td>
<td>0.621</td>
<td>0.167</td>
</tr>
<tr>
<td>Shanghai</td>
<td>0.091</td>
<td>0.091</td>
<td>0.167</td>
</tr>
<tr>
<td>Singapore</td>
<td>0.253</td>
<td>0.253</td>
<td>0.167</td>
</tr>
<tr>
<td>Sydney</td>
<td>0.825</td>
<td>0.825</td>
<td>0.167</td>
</tr>
<tr>
<td>Tokyo</td>
<td>0.491</td>
<td>0.491</td>
<td>0.167</td>
</tr>
</tbody>
</table>

¹For Scenario A of wastewater recycling, US$0.050 per m³ of the cost of treatment is energy cost, and for Scenario B, US$0.017 per m³ is energy cost.
by Sydney, then Miami. Not only do these cities have low population densities, they are also in the
developed world where individual cost components are relatively more expensive.

The cost of wastewater recycling, on a per unit basis, under the assumptions of this study, is uniform
across all cities under Scenario A, where treatment cost makes up most of the overall cost. Under Scen-
ario B, however, there is greater heterogeneity across cities due to the additional cost of constructing a
separate network of mains for carrying recycled water that is not present under Scenario A. For all cases,
wastewater recycling as represented by Scenario B is the more expensive option, though the difference
between the two scenarios reduces as population density increases. For instance, for Chennai, which has
the highest population density of all the cities examined, the cost of wastewater recycling under Scenario
B is only US$0.045/m³ more than under Scenario A.

The results in Figure 1 suggest that when compared to Scenario A of wastewater recycling, the
threshold of population density above which seawater flushing is likely to be economical is about
16,000–18,000 persons per km². This threshold is by no means definitive, however, given that popu-
lation density is not the only factor affecting the cost of seawater mains, which is also influenced by
purchasing power parity. Thus, even where the population density is smaller, it is still possible for sea-
water flushing to be favourable if the purchasing power parity is such that construction costs are
relatively cheap, which tends to be the case in developing countries. Similarly, seawater flushing is unli-
kely to be favourable where population density is higher but the purchasing power parity is such that
construction costs are also higher.

Figure 2 shows the potential savings from using seawater for toilet flushing as compared to using
recycled water. Results are presented for both Scenarios A and B for wastewater recycling given a recy-
cling plant capacity of 300,000 m³/day, as assumed above. For each scenario, two sets of results are
presented, one for traditional metal seawater and recycled water mains, and the other for PVC mains.
The results in Figure 2 and hereafter are expressed as ranges to account for possible overestimation or underestimation of costs. The ranges are derived by taking the costs of mains in Table 2 and applying the rule of thumb of adding $\pm 30\%$ to capital costs, and by adding $\pm 50\%$ to the treatment costs in the table, as found by Wittholz et al. (2008) to be more representative of desalination cost prediction uncertainty.

Fig. 2. Potential savings in 2010 dollars from seawater for toilet flushing over Scenarios A (left) and B (right) of wastewater recycling, and likelihood scores of the potential savings, assuming traditional metal (●) or PVC (●) mains.
For each estimate of potential savings, a likelihood score is assigned. A score of 1 suggests it is highly unlikely that seawater flushing is cheaper than wastewater recycling; a score of 2 suggests that it is possible but not likely; 3 is possible with a greater likelihood; and 4 very likely. A score of 1 is assigned when 0% of the range of potential savings falls above zero, 2 when 0.1–50%, 3 when 50–99.9%, and 4 when 100%.

From Figure 2, it can be seen that seawater flushing has greater potential for cost savings when compared to Scenario B of wastewater recycling than when compared to Scenario A, which is expected considering Scenario B incurs a higher cost because it requires the building of a separate network of mains. Similarly, seawater flushing has greater potential when PVC mains are used, instead of metal mains, due to PVC having a lower cost.

Three cities stand out for seawater flushing having entirely positive ranges of potential savings compared to both Scenarios A and B of wastewater recycling, assuming PVC mains (in other words, with likelihood scores of potential savings of 4): Chennai, Mumbai and Shanghai. Even assuming metal mains, their likelihood scores are at least 3, whether comparing with Scenario A or B. (For Chennai, for metal mains, the likelihood score is 4 comparing with Scenario B.) These results imply seawater flushing to be likely an economically viable (albeit unconventional) option for these cities, that should be seriously considered by policymakers.

Also, for Buenos Aires, Hong Kong, Jakarta and Karachi, there is a strong possibility of seawater flushing being economically viable. For these cities, for all combinations of wastewater recycling scenario and pipe material, the likelihood scores of potential savings are at least 3. For all four of these cities, the likelihood scores are at the maximum of 4 when compared to Scenario B of wastewater recycling for the case of PVC mains.

For other cities like Los Angeles, Miami, New York City, Osaka, San Francisco, Sydney and Tokyo, under Scenario A, there is almost no possibility of seawater flushing being economically advantageous regardless of pipe material. Under Scenario A, for these cities, the likelihood scores of potential savings are either 1 or 2. However, under Scenario B, the likelihood scores increase to either 2 or 3 depending on the pipe material assumed, which implies a stronger but still limited possibility of seawater flushing being a feasible option.

The overall cost of seawater flushing for a city, compared to that of wastewater recycling, is less sensitive to changes in energy cost. Apart from the energy to distribute water to households (which is about the same whether it is for seawater or recycled water), for wastewater recycling, additional energy is required for water purification. Thus, any increase in energy cost is to the advantage of seawater flushing but to the disadvantage of wastewater recycling.

Figure 3 shows the expected changes in potential savings from seawater flushing given increases of 25% or 50% in energy cost. The results are for the case of PVC mains and in comparison with Scenario A of wastewater recycling. (Results are shown for Scenario A, instead of Scenario B, as it leads to more conservative estimates of potential savings.) As expected, for all cities, potential savings increase with energy cost. However, except for Jakarta, Karachi and Miami, there is no change in the likelihood scores of potential savings. For Jakarta, the likelihood score increases from 3 to 4 as energy cost increases by 50%; for Karachi, it also increases 3 to 4; and Miami, from 1 to 2.

The results presented thus far assume that the per capita rate of flush water consumption of a city is the same as Hong Kong’s. However, this is not always true due to water conservation measures or differences between flush technologies. Figure 3 shows the sensitivity of the potentials of seawater flushing for the different cities to per capita rate of flush water consumption. Results are presented for 25 and 50% reductions in the rate of flush water consumption. Generally, the potentials for savings decline
as the per capita rate of flush water consumption reduces. However, in terms of the corresponding likelihood scores of potential savings, the declines in potential savings are inconsequential. This is true except for Osaka, San Francisco, Shanghai, Singapore and Tokyo, all of which show declines in their likelihood scores, though of no more than 1, with the reductions in the per capita rate of flush water.

5. Conclusions and future work

Seawater for toilet flushing is a resource that offers an unlimited and stable supply of water. However, it requires the construction and maintenance of a dedicated network of seawater mains, the costs of which may be prohibitive. This study estimates the unit costs of seawater flushing for 15 major coastal cities, Buenos Aires, Chennai, Hong Kong, Jakarta, Karachi, Los Angeles, Miami, Mumbai, New York City, Osaka, San Francisco, Shanghai, Singapore, Sydney and Tokyo. This study also estimates the unit costs of wastewater recycling for the cities. The study finds Chennai, Mumbai and Shanghai to have strong potentials for seawater flushing, compared to wastewater recycling. These cities have among the highest population densities in the world, and are located in the developing world and thus have relatively lower construction costs. Seawater flushing is also possibly economically viable in Buenos Aires, Hong Kong, Jakarta and Karachi. (In Hong Kong, it is already being practised and currently covers 80% of the population.) However, for cities like Los Angeles, Miami, New York City, Osaka, San Francisco, Sydney and Tokyo, due to their low population densities and relatively higher construction costs, seawater flushing is estimated to have zero or limited potential.
It has been estimated, at current prices and costs, that the critical value of population density above which seawater flushing becomes economically favourable to be about 16,000–18,000 persons/km². This estimate of the critical value is just a rough one, however, as the economic viability of seawater flushing depends also on purchasing power parity. Depending on purchasing power parity, a city with a population density less than the above estimate may still find seawater flushing to be favourable, or one with a greater density may still find it non-favourable.

Seawater for toilet flushing is unconventional, but as the results of this study shows, should be given serious consideration by policymakers searching for new sources of water to supplement conventional surface and groundwater resources that, in many places, are reaching their limits. This is especially true in light of the rapid urbanization of many of the world’s populations that is inevitably leading to higher and higher population densities and, therefore, to lower and lower unit costs of seawater mains. (According to the United Nations (2012), the total population in urban areas is projected to increase from 3.6 billion in 2011 to 6.3 billion in 2050, much of which is expected to be concentrated in the developing world, specifically Asia and Africa.)

Seawater flushing should also be considered for its potential savings in carbon emissions. Compared to wastewater recycling, seawater flushing consumes significantly less energy and thus emits considerably less carbon. If a positive value is assigned to carbon savings, seawater flushing may possibly be the optimal choice for many more cities, and not just the ones identified in this study. A systematic assessment of the potential for reductions in carbon emissions of seawater flushing is beyond the scope of the present study, and left for future work.

Wherever seawater flushing is adopted, care should be taken to avoid any discharge of treated or untreated saline sewage to inland waters to prevent any risk of intrusion of saline water into groundwater or any possible damage to freshwater ecosystems. Instead, the sewage should be discharged to the sea. Further, any decision to adopt seawater flushing should take into account possible long-term consequences on the ecology of receiving marine waters. One possible consequence is the effects of brominated disinfection byproducts formed when chlorinating saline sewage effluent (Ding et al., 2013). These byproducts are not typically found in chlorinated sewage that is non-saline, and their toxicities to marine life are yet to be fully understood. Similarly, other ecological consequences of seawater flushing are not yet well fathomed and require additional research to identify and quantify them. It is possible that, once understood, the ecological consequences of seawater flushing may limit its applicability. The recommendations made in this paper do not consider these consequences and are based solely on cost considerations.

Acknowledgements

This study is supported, in part, by grant 617012 of the General Research Fund scheme and grant DAG12EG02 of the Direct Allocation Grant program, both from the Research Grants Council of Hong Kong.

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Received 15 March 2013; accepted in revised form 8 June 2014. Available online 30 June 2014.