Carbon sensitive urban water futures

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TRANSITIONS TO THE URBAN WATER SERVICES OF TOMORROW

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Final Version for Distribution

April 2012

The research leading to these results has received funding from the European Union Seventh FrameworkProgramme (FP7/2007-2013) under grant agreement n° 265122.

EXECUTIVE SUMMARY

The challenge of supplying water and energy required for food production and development while mitigating climate change and adapting to its consequences has been termed the Energy Water Nexus. Water, energy, greenhouse gas emissions and climate change are interlinked through a series of relationships. This document seeks to respond to and further develop the concepts, opportunities, and agendas needed to drive more energy and carbon sensitive water management from a European perspective. Our findings illustrate that the water supply and wastewater industries throughout Europe are significant consumers of energy and emitters of greenhouse gases but have ambitions to make emissions reductions of 20% by 2020 and 80% by 2050.

Our exploration of strategies for achieving such reductions suggest that demand reduction, through the reduction of waste and increasing water efficiency at the point of use offers savings of up to 10% by 2020 and at least 20% by 2050. Progressive improvements in the efficiency of conveyance and distribution should also be able to make a positive impact with improvements to motor and pumping efficiency within water treatment works reducing energy use by a further 10–45%. Other operational efficiency improvements have been shown to reduce the total energy consumption by up to 40% in some cases. Replacement of GHG-intensive treatment processes, such as GAC filtration, has also saved up to 40% of emissions. However, the savings potential for the water supply sector probably falls short of the 80% emission reduction target for 2050. This is especially true if source restrictions force the use of lower-quality supplies, such as recycled, brackish or desalinated water. The energy cost of treating these is falling, but is still generally higher than freshwater.

In wastewater treatment we find that improvements to the efficiency of pumps and other motors could provide efficiency gains of 10–40% and other general operational improvements may be capable of saving up to 25% of total plant energy. Aerobic treatments, especially the activated sludge process should be a focus for more substantial reductions. Better process control and other efficiency measures, some of which might be combined, have been shown to reduce consumption by 10–50%. These should enable reductions of 20% of total emissions in the short to medium term. Longer term savings are likely to require the replacement of the main aerobic treatment processes by anaerobic treatment. Biogas from anaerobic digestion of sewage sludge is an important source of renewable energy and should be maximized by ensuring that it is captured and used for combined heat and power generation. A move to anaerobic primary treatment would reduce plant energy requirements and increase biogas production, providing a net energy export. Finally, dewatered or dried sludge used as a fuel (perhaps via co-firing) shows promise as a way of off-setting treatment energy use.

Of equal importance however is the knowledge and professional culture required to drive innovation and delivery of low carbon and energy solutions. Our work suggests that this, perhaps above all other factors, is the area in need of catalysis if our communities are to realize a lower carbon future.

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1. INTRODUCTION & CONTEXT

The passing of the centuries has not changed human beings' ability to create. Yet as a result of technology, today's customs and lifestyles bear little resemblance to those of just fifty years ago. The solid foundations of knowledge established by our predecessors began to bear fruit in the industrial revolution, in the second half of the 18th century. But nothing can compare to the technological advances of recent decades which have led to the globalisation that currently holds sway. Everything has happened so fast that our present way of life does not resemble that of even a few decades ago. Contrast this with people who lived in the 10th century. Had they been born two centuries earlier, they would have noticed little change.

These advances have taken place across a range of engineering fields, although a distinction should be made between those with a short, as opposed to a long, history. The former (e.g. telecommunications or aeronautics) have evolved in lockstep with emerging socio-economic frameworks. This is not the case with water engineering, a form of engineering with a long and influential history as is witnessed by the magnificent structures that have been left to us (Cabrera and Arregui, 2010); built in order to meet a basic need. As a result the technological breakthroughs of the 20th century were obliged to coexist with an almost ancestral culture and a rigid and consolidated governance system, and this has led to significant disparities. The reasons are twofold: firstly, dovetailing rapid technological and social change with ancient laws and rights is a complex task, and secondly this kind of rigid framework makes adapting water policy to contemporary circumstances difficult, especially in countries where the history of water is of great importance (Bru and Cabrera, 2010). However, rectifying disparities is easier in the case of new technologies since progress and frameworks move virtually in tandem.

In short, a culture of water use which is as subsidised as it is entrenched makes it hard to tackle current problems that are caused largely by very fast and unbalanced technological development. Consequently the structures of the institutions tasked with regulating water management, the training of decision-makers and the cultural attitudes of the public at large are inadequate to deal with the collateral damage of progress. The reforms that are required therefore need to be undertaken as a matter of urgency since the problems will grow with time. Furthermore, if the complexity of the reforms continues to alarm those who have to carry them out, then the outcome is all too predictable; a major crisis creating additional and more immediate challenges.

The current imbalance began in the 20th century with the tremendous development of civil, hydraulic and electromechanical engineering. Until then, societal development had taken place much more gradually with problems and their solutions going practically hand-in-hand. However, huge dams and pumps changed the dimensions of hydraulic engineering as it became possible to store large volumes of water and transport it over hundreds of miles. This development enabled the achievement of goals that previous generations could only dream of, as millions of hectares of dry land were irrigated and hitherto uninhabitable places (such as Las Vegas) were settled.

Water engineering had achieved its greatest standing, allowing Rouse to affirm that "*Hydraulicians are human too*" (Rouse, 1987). Yet ironically he wrote these words just when the expansionist water policies of the 20th century peaked. In the very same year, the Brundtland Commission, concerned about increasing environmental deterioration worldwide, presented its findings (UN, 1987). Since then, the only water policy that humans had previously implemented (making more resources available) has found its counterweight in a demand management paradigm that is more committed to efficiency. Yet, because this latter goal has so far been pursued somewhat timidly (mainly in semi-arid countries) and not across the board, the mismatch between supply and demand remains. Inertia hinders progress (Sheer, 2010) because the solution chosen first by change averse individuals, is "business as usual"*.* Only enhanced environmental education for society, currently thin on the ground, can counteract the weight of history in terms of these problems.

1.1. The new context of urban water planning in the 21st century

Supplying the world's growing population with quality water is one of the biggest problems facing society today because it affects people's quality of life and, even more, their survival. It is a particularly complex challenge in an urban context due to:

Extraordinary and asymmetric population growth

Over the past six decades the world's population has nearly tripled (Figure 1). While in 1950 the earth had 2.5 billion inhabitants, today there are 7 billion of us, an impressive increase given that in the previous nine centuries the rise was "only" 1.4 billion (from 300 million at the start of the second millennium up to 1.7 billion at the beginning of the 20th century). And although the population growth slope has become more shallow, the ordinate is increasing so that by 2050 the world's population will have reached 9 billion people. If this forecast turns out to be correct, the Earth will have played host to an extra 6.5 billion people in the course of a century.

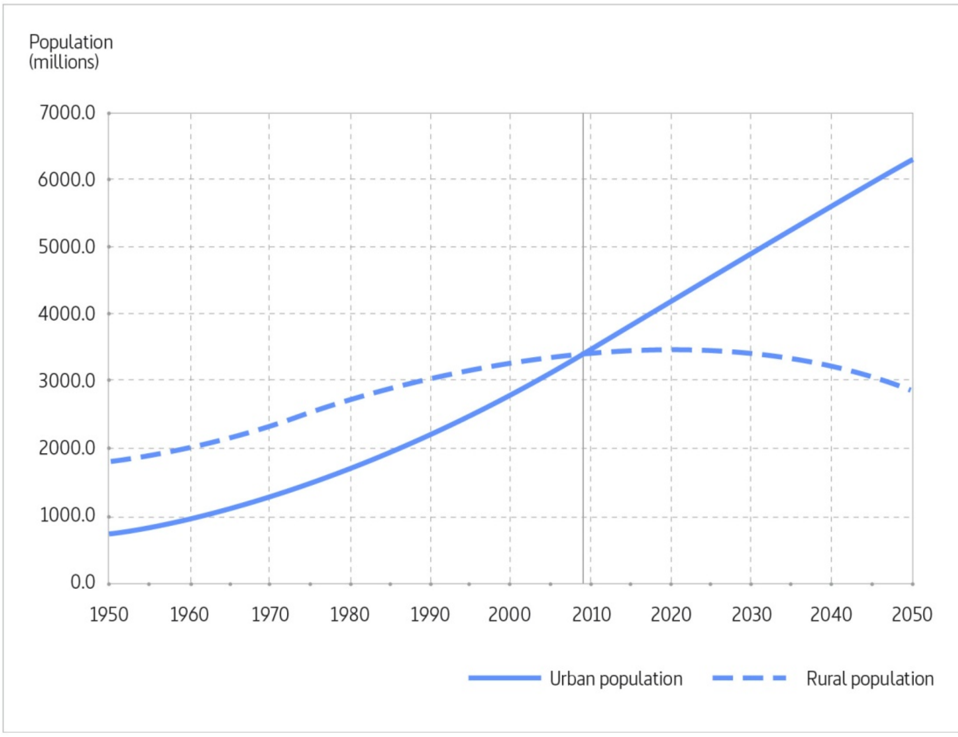

Figure 1: World Urbanization Prospects (UN, 2010a)

Yet the figures are even more striking if comparisons are made in terms of urban population. In 1950 only one third of the inhabitants of the Earth (700 million people) lived in cities. By mid-2009 there were already 3.4 billion urban dwellers, accounting for 50% of the world's population. Furthermore it is estimated that a century later in 2050 this figure will have reached two thirds of the population (UN, 2010a). In short, in just one hundred years cities will have gone from 700 million to 6.3 billion inhabitants. Nothing else really needs to be said, mainly because millions of people lack access to drinking water which has prompted the UN resolution stating that water is a universal human right (UN, 2010b), a statement that does not solve the problem but does bear out the saying that *Excusatio non petita, accusatio manifesta*.

Water needs are increasing significantly in an uncertain scenario dominated by climate change

At present agriculture uses 70% of water resources (Molden, 2007) although forecasts indicate that, in absolute terms, agricultural use will decrease slightly. In relative terms, the decline will be more significant (in 2050 it will account for 50%) because domestic and especially manufacturing (in some way also domestic) use will increase considerably (Figure 2). These forecasts were undertaken prior to the food crisis of 2008. Since then it is believed (Ludi, 2009) that as food production will need to increase substantially, water supply for irrigation will also rise moderately in 2050, adding more pressure to this valuable resource

It is true that some developed countries are improving water efficiency and promoting the use of alternative resources (greywater and rainwater) by means of implementing charges that recover costs, thereby reducing unit demand (WVGW, 2008; Rockaway et al., 2011). However, since this is not a widespread policy, the efficiency (the key to the future) of a few does not outweigh the increase in population and water use in other regions. Furthermore, water demand is likely to continue to grow in lockstep with the rising population, leading to further strain on water resources due to growing demand and dwindling availability in a context of climate change, particularly in those countries where water is already in short supply (Milly et al., 2008).

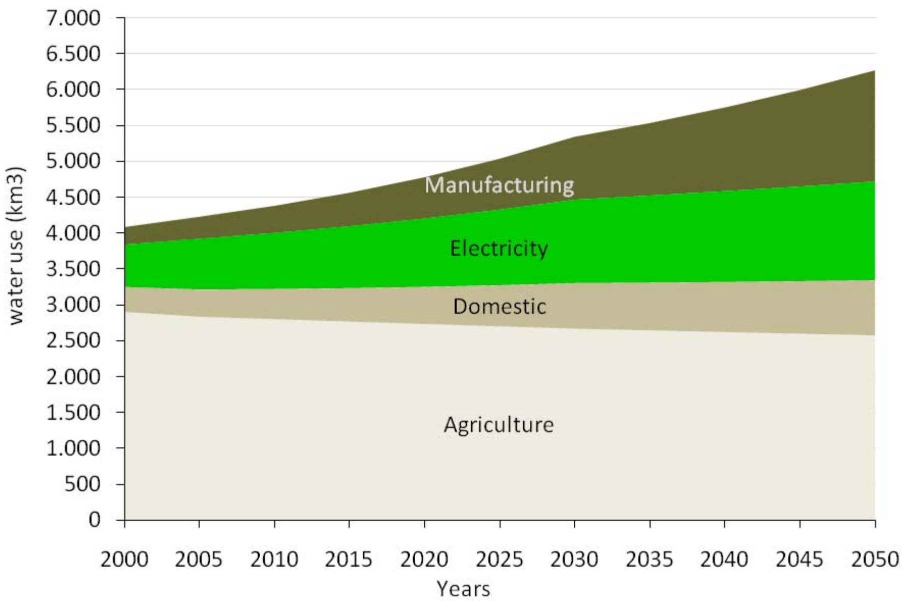

Figure 2: World Water Use by Sector Prospects (OECD, 2008a)

Urbanisation and land use change continues to impact water resources.

Land use and environmental sustainability are an inseparable pairing (Lambin and Meyfroidt, 2010), which explains the growing interest in them. In the present case, the urban development of cities has a very direct impact on and increases the complexity of sustainable water management in the human environment. The creation of impervious surfaces adds to runoff, reduces the recharging of aquifers and gives rise to urban flooding (NRC, 2008). Its increasing frequency and the high economic and, occasionally, human damage it entails have made this issue one of the greatest concerns of some cities. Other land use changes, especially deforestation, affect the hydrological cycle and special attention is being paid to the Amazon region due to its great importance (D'Almeida et al., 2007). Changes in land use, especially if they result in reduced plant cover (Sugden *et al.,* 2008), has a direct influence on both water supply (through hydraulics) and demand (through newly urbanised areas). Land use change and water management are indeed so closely related that it is fair to say that town and country planning involves setting water policy (Falkenmark and Rockström, 2006).

Water pollution is steadily increasing

Another notable change over recent decades has been the deterioration of quality of the water people use. While in the past irrigation did not bring with it toxic substances, the use of fertilizers and pesticides has contaminated many of the aquifers that supply urban areas. These aggressive practices began in the second half of the 20th century and in Europe the reaction came in 1991 when the Nitrates Directive was enacted to become one of the first instances of environmental legislation in the European Communities (EC, 1991). It marked a turning point in this field and has helped to improve the situation slightly (EC, 2010). Yet it remains one of the issues that cause most environmental concern (OECD, 2008b).

Nevertheless urban (including storm water collection) and industrial water uses, due to the pollutants they bring with them, have the highest impact on water quality. Furthermore, the quantity of water polluted and accumulation in aquifers is permanently growing due to population increase, urbanisation, and growth in unit consumption (litres per person per day). Thus, between 1950 and 1990 these flows tripled (Turner II *et. al.*, 1990) while the population only doubled. Restoring natural water bodies to an acceptable quality calls for major investments and considerable operational costs that many developing countries are unable to afford. Although much depends on the size of the plant and the processes used, the cost of treating water (including investment and operations) is about €0.30/m^3 (Torregrosa, 2010). Here one of the biggest costs is energy, which explains why one of the key factors for water and energy is the energy efficiency of wastewater treatment plants (GWRC, 2008).

The current economic crisis

We are living in turbulent times with few developed countries, if any, not affected by the global financial crisis. One of the biggest problems is the impact which excessive government borrowing is having. Nation states across Europe need to reduce their deficits, meaning that in forthcoming years they will be faced with tight budgets. This will hamper what has hitherto been standard practice in many countries; paying for infrastructure with public money. To put it another way, subsidies will gradually be withdrawn. Indeed, since 2010 they have been banned in Europe by the Water Framework Directive (EC, 2000) although many countries, especially against the background of the current economic crisis, have been struggling to put this legislation into practice.

The probable end of subsidies, in addition to leading to an increase in prices, will make it necessary to seek alternative funding for work that cannot be put off and to reduce costs. The need to attract private capital will reopen the perennial debate about public-private management (Boland, 2007) which, irrespective of the pros and cons of each model, is a subject that should be discussed from a pragmatic standpoint isolated from the political arena. In addition, the need to reduce costs will foster efficiency and make it necessary to seek out economies of scale. As a matter of fact, just in the USA, can be found 150,000 organizations related to urban water (Grigg, 2007). Services will have to be pooled, a strategy that some countries such as Italy began with the Galli Act (GU, 1994) a number of years ago and which now, by abolishing provinces and municipalities with only a few inhabitants, is being extended to all services. The crisis predominates. It is a strategy that some countries such as the UK had very much in mind when they privatised the service by delegating management right from the off to a very small number of operators. By then, there was a vision of the future.

In short, the current economic crisis will in all probability mark a turning point in terms of subsidies and the recovery of water costs. It must be pointed out that in those countries belonging to the euro area, in which the economic crisis is deeper (Greece, Portugal, Ireland, Spain and Italy), the water sector is significantly subsidized. In the medium to long term, there will be strong drivers for this situation to change and water prices are likely to rise progressively.

The challenge to maintain investment levels

The 20th century, particularly in its early decades, witnessed the building of large water engineering infrastructures. As part of a water policy geared towards mobilising more water resources, water engineering structures were an excellent driving force for an economy which had to be got going after the Great Depression and World War II (NAS, 2004). Furthermore, at that time government was not burdened by current debt levels and these were popular projects that enhanced quality of life and were, therefore, political vote winners. And since their environmental impacts were not well known, no one objected to them.

However, that is now all part of history, especially in developed countries where water management needs to take precedence over water development (Burgi, 1997). The water engineering structures of the past have to be preserved if not replaced, including millions of miles of urban pipelines. Yet this is not an attractive target for investment to the public at large who attach little value to it, unless they know it to be necessary. Nor is it attractive to politicians; renewing pipelines is often viewed as “burying” money. However, since the need is obvious so as not to further jeopardise future generations (Copeland and Tiemann, 2010; ASCE, 2011), it cannot be put off.

1.2. The great challenge: Improving resource efficiency.

Taking into account the preceding facts, the conclusions of the last stakeholder consultation organised by DG Environment of the EC to identify water innovation priorities (February 1st, 2012) cannot be a surprise. One sentence summarizes the achieved results *“Resource efficiency will be key for all countries, mainly the developing ones, and in particular for energy and water. It will open new markets and opportunities for the European water sector”.* These conclusions are in tune with one of the top priorities of the current Danish Presidency, to accelerate the transition to a low Carbon Economy. The European Union has recently defined a Roadmap in support of this ambition (EC, 2011a) and has announced a directive on energy efficiency (EC, 2011b). Most large-scale energy conversion processes consume water while sustainable urban water management requires significant amounts of energy. Both resources are thereby strongly coupled. As will be argued below, a new and integral approach is required to manage both resources properly, a strategy that gives rise to challenges and opportunities of innovation and research.

Traditionally, interest in optimising the use of energy has depended just on its price. A strong correlation is evident between budgets devoted to R+D and the price of the oil per barrel (IEA, 2008). The comparison highlighted a strong but slightly delayed correspondence (Figures 3 and 4). In particular the 1973 crisis can be easily identified. More recently the momentum for energy efficiency has not only been driven by financial considerations. R&D expenditures have also been positively influenced by the desire to reduce greenhouse gas emissions. Over the last five years (not displayed by these two figures) resources devoted to R+D and oil prices have not been so coupled as Figures 3 and 4 show.

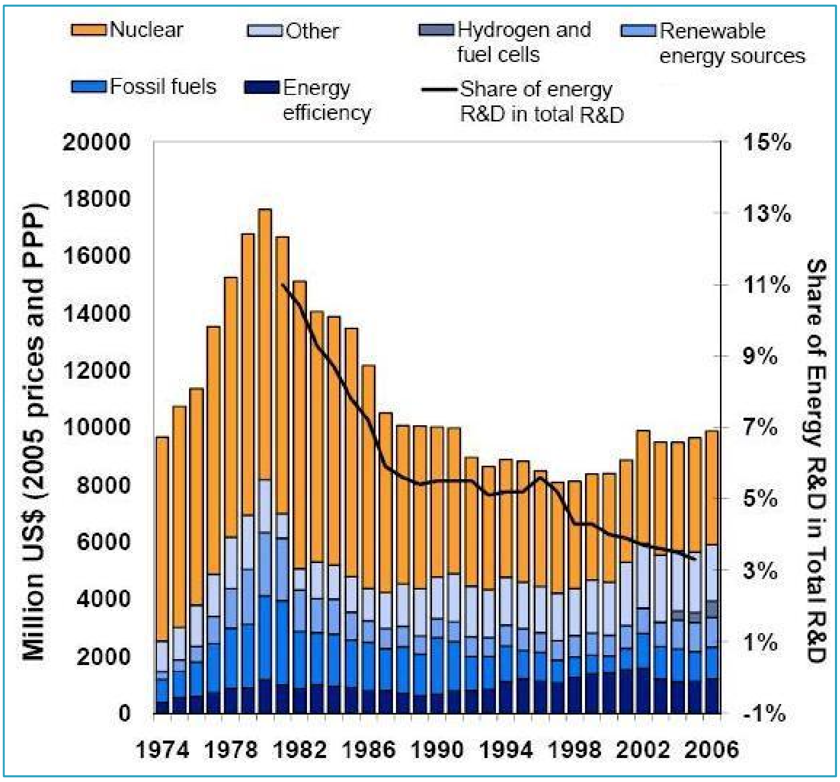

Figure 3: R+D investments in the IEA countries (IEA, 2008)

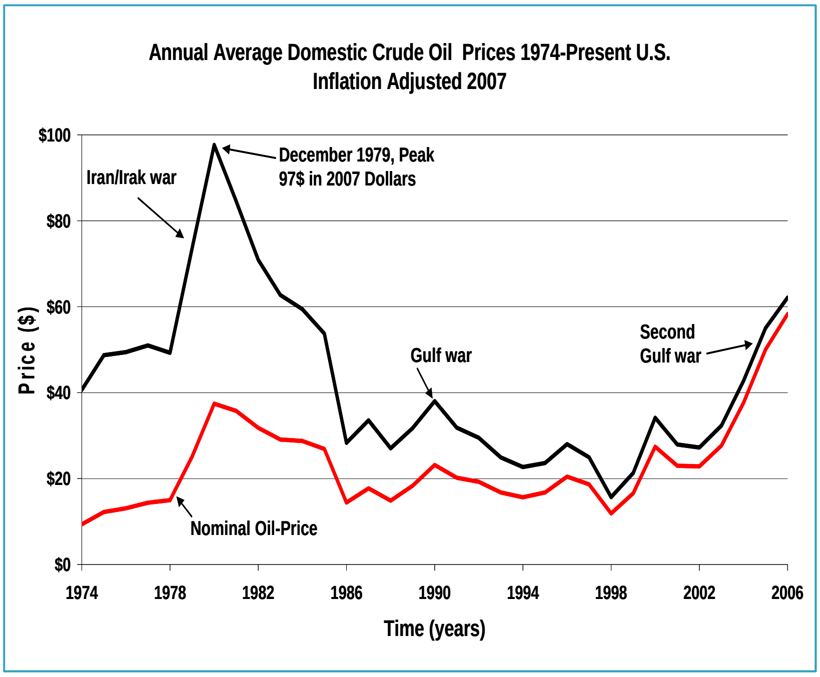

Figure 4: Oil prices in 2007 USA$ (www.inflationdata.com)

Since the first valuable report on the water energy nexus was released by the California Energy Commission in 2005 (CEC, 2005), interest in this issue has increased all over the world dramatically. In the CEC report, the energy footprint of the various steps of the water cycle were calculated, evidencing that urban and agricultural cycles have their own energy requirements. Once the energy footprint of the cycle has been determined, a sensitivity analysis of the different strategies can be performed in order to identify the more convenient approaches to save water (and then, energy as well). Figure 5 shows the range of the energy footprint (for each step) corresponding to the water cycles analysed in California. The difference between the least and greatest value is, in some phases of the cycle, important. For instance, at the water treatment stage, the minimum value corresponds to a clean natural source of water while the maximum is the energy required by a desalination plant.

From the volumes of water consumed by the different urban and agricultural cycles, the total amount of energy required is calculated. In total 48 GWh of electricity demand, an impressive figure that represents 19% of the State's consumption (250 GWh). This percentage rises up to 32% of the gas demand. All in all justifies that the US Congress showed great interest requesting to the Department of Energy a study on the water – energy nexus (USDE, 2006).

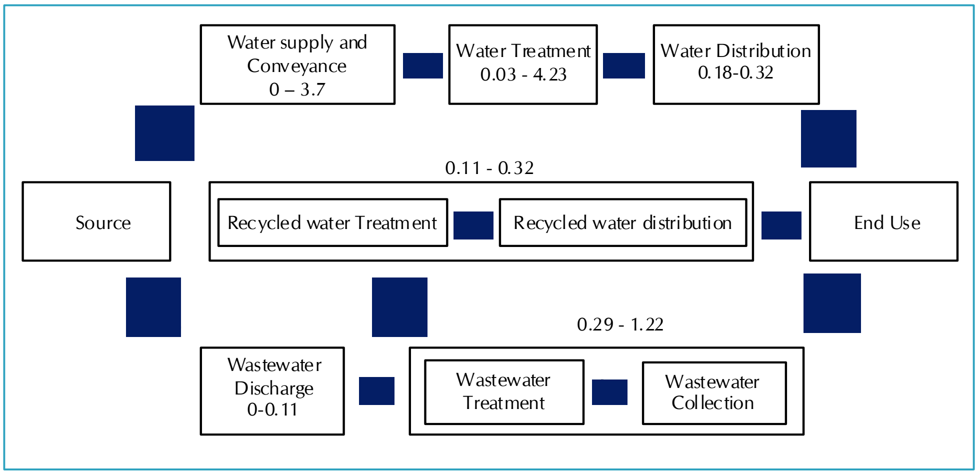

Figure 5: Water cycle energy requirements (kWh/m³) in California (CEC, 2005).

This study supports evaluation of energy efficiency potentials. Three strategies can be considered. First to optimize water transport, diminishing friction losses and improving the efficiency of pumping stations, and/or improving water and wastewater treatment processes, a strategy that is being explored in countries that already have efficient water management systems (GWRC, 2008).

A second possibility is to diminish the volumes of water being treated and transferred. If less water is mobilised, the energetic requirements drop significantly. This objective can be achieved by two ways. First by reducing water demand and secondly by minimising water losses (leaks) particularly in water distribution networks. This second option has been analysed in several reports and papers by one of the current author. In Cabrera *et. al.*, (2010), a metric that correlates water and energy losses is established. And secondly (Cabrera et al., 2009), water losses are linked to greenhouse gas emissions. For this purpose the Pacific Institute (Wolff *et. al.*, 2004) developed an excel sheet that correlates used energy with mass of CO_2 emitted. Other contaminants (carbon monoxide, nitrogen oxides, sulphur oxides, etc.) can also be included in the analysis.

The third strategic option is water reuse (to be discussed in more detail below) which in most cases, but not all (Rozos et al., 2010), saves energy as well. Reuse and recycling does, of course, bring additional benefits but energy efficiency has been flagged up as a potentially significant consideration here.

These opening discussions have set the context and articulated the challenge facing our professionals and communities as they seek to achieve low carbon and low energy water systems. Recent research in both Australia (Kenway *et. al.*, 2008) or Canada (Mass *et. al.*, 2011), provide tantalising glimpses of the potential for a carbon sensitive urban water future. The remainder of this document seeks to respond to and further develop the concepts, opportunities, and agendas from a European perspective. In the following sections we characterise current energy and carbon use profiles in the industry and explore the potential of new technologies and innovations to deliver on the ambition of a low carbon future.

2. THE ENERGY – WATER NEXUS AS A CONTEMPORARY CHALLENGE

As noted above, the challenge of supplying water and energy required for food production and development while mitigating climate change and adapting to its consequences has been termed the Energy Water Nexus (Hoyle, 2008; AAAS & JRC, 2011). Water, energy, greenhouse gas emissions and climate change are interlinked through a series of relationships. Most forms of generation of electricity consume water: 1–4 m^3/MWh for conventional thermal generation (WssTP, 2011). Water supply and wastewater treatment require energy, particularly electricity. The water industry emits greenhouse gases indirectly, through energy use, and directly, by releasing methane and nitrous oxide (N_2O) during treatment, so contributing to climate change (see Figure 6). One effect of climate change in many parts of the world is likely to be a change in rainfall patterns, which may further restrict the supply of groundwater and surface water in areas where it is already limited and even reduce availability in comparatively water-rich countries, such as the UK (Warren & Holman, 2012). At the same time, many countries have increasing populations and demands for water. This could create positive feedback by driving the use of other sources, such as desalination of sea water, which may have higher energy demands and greenhouse gas emissions.

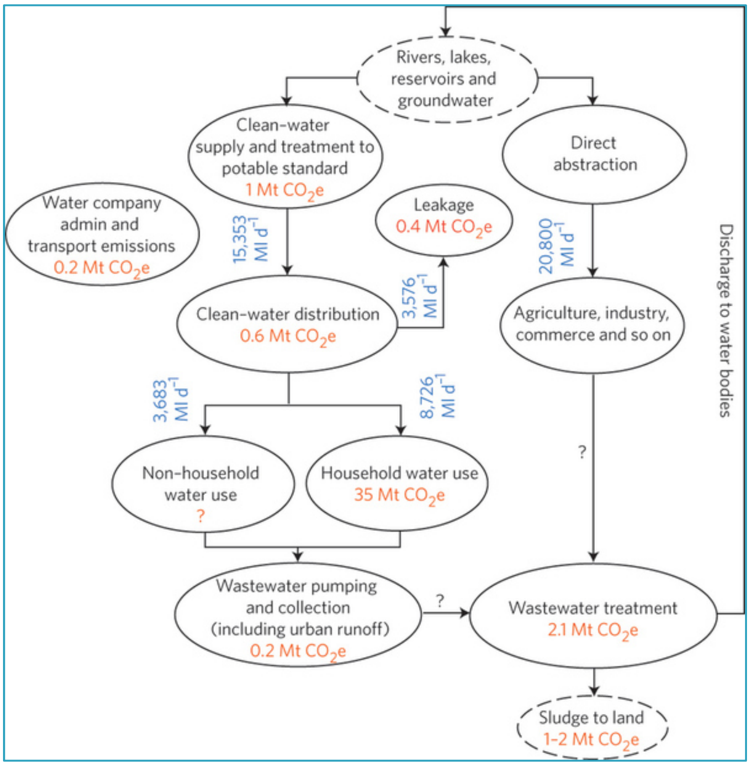

Figure 6: Water cycle GHG emissions from UK water sector 2005-6 (Rothausen & Conway, 2011)

Within Europe and elsewhere, the aims of different pieces of environmental legislation can come into contact. For example, the Water Framework Directive (WFD) requires high environmental quality standards for surface water bodies, further raising the energy intensity of wastewater treatment. In the longer term, the WFD may result in higher quality source waters and reduce the energy requirement for drinking water treatment. However, the effectiveness of the catchment management measures currently being taken to deal with diffuse agricultural pollution is uncertain (Smith & Porter, 2009; Cook *et al.*, 2011). The residence times of some groundwater systems are several decades (Burt *et al.*, 2010), so the changes may be very slow to take effect. This report will review and elaborate on these issues as they relate to municipal water supplies, and consider some of the options for sustainable water systems in the future.

The water sector is a significant contributor to greenhouse gas (GHG) emissions through several routes including embedded emissions in capital equipment, energy consumption during drinking water treatment, water distribution and wastewater treatment, and direct emissions of methane and nitrous oxide from treatment. An estimate that 5.5% of the UK's GHG emissions are associated with water has been widely quoted in the UK and abroad (Reffold *et al.*, 2008). However, this is misleading when taken out of context, because it includes a large component due end use, such as to domestic water heating. It is estimated that 89% of the GHG emissions are associated with the end use and that the operational emissions from the industry were 5 Mt CO_2e, or 0.8% of the national total (Reffold *et al*., 2008; Water UK, 2007). In less temperate climates with more limited water sources, the energy requirements may be higher. The corresponding estimate for the US is 1–2% (Slaa, 2011) or 4% (Means, 2003) and for California 7–8%, of which supply pumping is the largest component (Means, 2003, ACWA, 2011). Further evidence on the energy intensity and carbon footprint of supply and wastewater treatment will be considered below.

The extent of the impact of any changes in the water industry on national greenhouse gas emissions in most European countries is, therefore, likely to be very limited. However, it expected to reduce emissions in line with the EU Climate Change Package and national targets, which form a legal obligation in the UK, while continuing to meet the demand for high-quality water supplies. The infrastructure of the water industry contains many assets with high capital costs and long lifetimes: 15–30 years for electrical and mechanical assets and 30–60 years for civil assets (Palmer, 2010). The populations and industries served may change more rapidly, so the industry has to be plan for varying demands and loads. As a result, it cannot respond quickly to external pressures.

2.1. Overview of the water-wastewater system

There are five main stages in the water-wastewater system, linked by transport steps (Figure 7). Water is collected and extracted from its source, treated to appropriate chemical and biological standards, used by consumers, returned in part to the sewerage system, which also collects drainage water from buildings, roads and other surfaces, treated to further chemical and biological standards, and discharged, usually to surface water bodies or the sea. The two treatment stages consume energy, as do the transport steps connecting them.

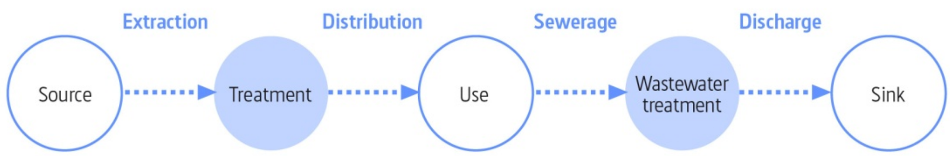

Figure 7: The five stages of the water-wastewater system.

(The stages with bold outlines are within the companies' control. Arrows are transport steps.)

The two treatment stages are transformative processes which take an input of variable quality and produce an output whose quality is strictly regulated. The input quality is partially outside the control of the water companies. It is influenced both by natural phenomena, such as rainfall, and the behaviour of other sectors of society, including agriculture (diffuse agrochemical pollution), industry (discharges of pollutants to surface waters and urban drainage) and consumers. The need to treat highly variable inputs and the long lifetimes of treatment assets means that the companies have to be conservative in the design of facilities, to ensure their ability to deal with worst cases. The source, extraction, treatment, and distribution processes are usually grouped into a water supply function. Similarly, sewerage, wastewater treatment and discharge are grouped into the wastewater function.

There are many different treatment processes applied to both water treatment and wastewater, with different energy intensities and levels of GHG emissions. For water treatment the processes used depend in part on the contaminants present in the water sources. They include filtration, oxidation, ultraviolet treatment, denitrification desalination and chlorine disinfection (Rothausen & Conway, 2011; Klein *et al.*, 2005).

Wastewater treatment typically starts with preliminary treatment to screen out sand and grit. Primary treatment then separates it into two streams, primary sludge and clarified effluent, by settling or sedimentation. The effluent undergoes secondary treatment, usually aerobic, to break down organic compounds, which gives rise to further sludge. The sludge is usually thickened and then either dried or used in anaerobic digestion. A tertiary step may be applied to the effluent to remove further organic matter or specific pollutants, or to disinfect the water (Slaa, 2011; POST, 2007; Palmer, 2010). The flowsheets in common use have changed little in the last century, but are now being reassessed (Stephenson, T. in Ainger *et al.*, 2009). A review of sludge production and treatment options in use across Europe was recently conducted as part of the FP7 project END-O-SLUDGE (unpublished interim report, personal communication). There were two sludge production systems, depending on the scale of operation, but both including screening, an activated sludge unit and thickening. There were three sludge treatment steps used in different combinations according to the disposal route: digestion (usually anaerobic digestion), dewatering and thermal drying. The four disposal options were fuel, farm use (as fertiliser), landfill and incineration. The four combinations of the treatment options found were

- dewatering only – for farm use, landfill or incineration
- digestion and dewatering – for farm use, landfill or incineration
- dewatering and drying – for fuel or farm use
- digestion, dewatering and drying – for fuel or farm use

A carbon footprint study for the UK considered a similar range of options. All included dewatering preceded by either two stages of digestion or pre-treatment followed by digestion. The disposal options were land application, drying followed by land application or co-firing, and incineration (Barber, 2009).

2.2. Legal framework for drinking water and wastewater treatment

Several EU Directives provide a legislative and regulatory framework for the European water sector. The Urban Wastewater Treatment Directive (UWWTD - 91/271/EEC) and the Drinking Water Directive (DWD - 98/83/EEC) significantly influence the level of treatment required for wastewater and potable water respectively; thereby influencing technology choice, energy use and GHG emissions. These influences are well documented with a prime example being the case of nutrient removal. Water operators across Europe have expressed concerns for several years now about the increased energy use (and associated carbon emissions) and chemical use needed to achieve nutrient removal rates required by the UWWTD. Phosphate removal by the use of ferric salts is an effective (and cost efficient) approach to achieving compliance in this area but as well as adversely impacting the chemistry and biology of the receiving water body such practices are relatively energy and materials intensive.

The other major piece of Europe-wide legislation to influence water management is the Water Framework Directive (WFD - 2000/60/EC) came into force in October 2000 (Art.25) and will become fully operational this year (2012). Its primary requirements are to prevent any deterioration in water quality in any water body and to aim to achieve 'good status' in all water bodies except those designated as an Artificial, or Heavily Modified, Water Bodies (Art.1 & Art.4).

The actions ('programme of measures', Art.11) required to achieve these objectives have been set out in statutory River Basin Management Plans(Art.13) for each designated River Basin District (Art.3). The actions include 'catchment management' interventions as well as 'end of pipe' treatment options.

The WFD requires that existing interventions under previous Directives (Urban Waste Water Treatment (91/217/EEC), Dangerous Substances in Water (76/464/EEC)[#], Pesticides (91/414/EEC), Nitrates (91/676/EEC), Bathing Water (76/160/EEC), Surface Water for Drinking Water (75/440/EEC)[#], Drinking Water (98/83/EC), Freshwater Fish Water (78/659/EEC)[#], Shellfish Water (79/923/EEC)[#], Groundwater (80/68/EEC)[#], Habitats (92/43/EEC), etc., continue to be implemented (Art.10 & Art.16). A number of these Directives[#] will be repealed once the WFD is operational (Art.22).

The Commission have stressed that if these existing Directives have been fully implemented effectively and on time, then water status should already be good and the need for any additional measures should be limited.

In the past, setting higher standards for discharges from wastewater treatment works has contributed significantly to improvements in the water quality of many rivers. It is expected that this mechanism will continue to help to deliver WFD objectives.

The increasingly high standards for discharges from wastewater treatment works have led to increasing energy use, which is estimated to have doubled in the UK between 1990 and 2011 (WssTP, 2011). This has led some authors to suggest the need to reconcile the demands of different pieces of environmental legislation, such as the WFD and the Climate Change Package (Baleta & McDonnell, 2011).

Regarding drinking water, the two primary objectives of the WFD, i.e. no deterioration and achievement of good status, should mean, initially and as a minimum, no increase in water treatment to achieve drinking water standards and eventually reduced water treatment as raw source water quality improves. Indeed this is explicit in Art 7, which can be regarded as an extending the requirements of Directive 75/440/EEC.

> Art.7.3 – "*Member States shall ensure the necessary protection of the bodies of water identified*" (under Art7.1, those used for the abstraction of water intended for human consumption) "*with aim of avoiding deterioration in their quality in order to reduce the level of purification treatment required in the production of drinking water.*"

This sounds straightforward but the difficulty for water companies, who need to plan ahead with their investment strategy for the necessary water treatment infrastructure to meet strict drinking water quality standards, is that in reality there is large degree of uncertainty over if and when WFD objectives will be achieved under the proposed programme of measures. With pesticides and nitrates being contaminants of particular concern in raw water sources the impact of those Directives under the framework of the WFD are also important but equally uncertain.

2.3. Water consumption and sources at national scale

The total volume of water extracted varies substantially between EU countries, even those with similar population sizes (Figure 8). These data are derived from FAO/aquastat (http://www.fao.org/nr/water/aquastat/main/index.stm), using the most recent data available for each country. The volumes extracted for municipal use are more consistent, but still vary considerably. The proportions extracted for use by the three main sectors – municipal, industry and agriculture – are highly variable (Figure 9). Many southern European countries (e.g. Greece, Spain, Italy and Portugal) extract large volumes for agricultural use compared with most central and northern countries, though agricultural use in some Scandinavian countries is also moderately large.

Note that, the dates when the data were collected vary by up to 10 years and some of the values are estimated or modelled by the FAO, so the error bounds are variable and unknown. Where primary data are used, there may be differences between countries in the method of collection and the allocation to different sectors.

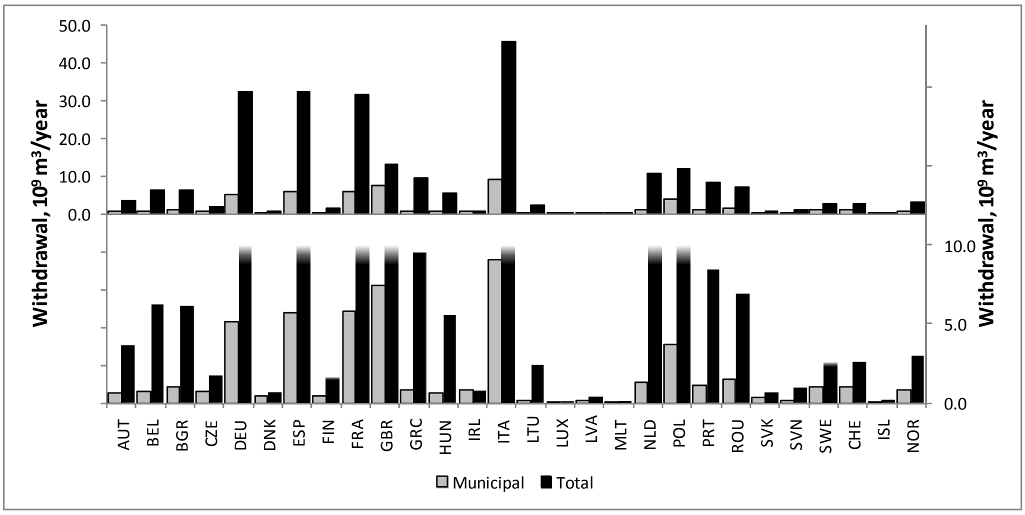

Figure 8: Annual municipal and total water withdrawals by European countries.

(Data from FAO/aquastat)

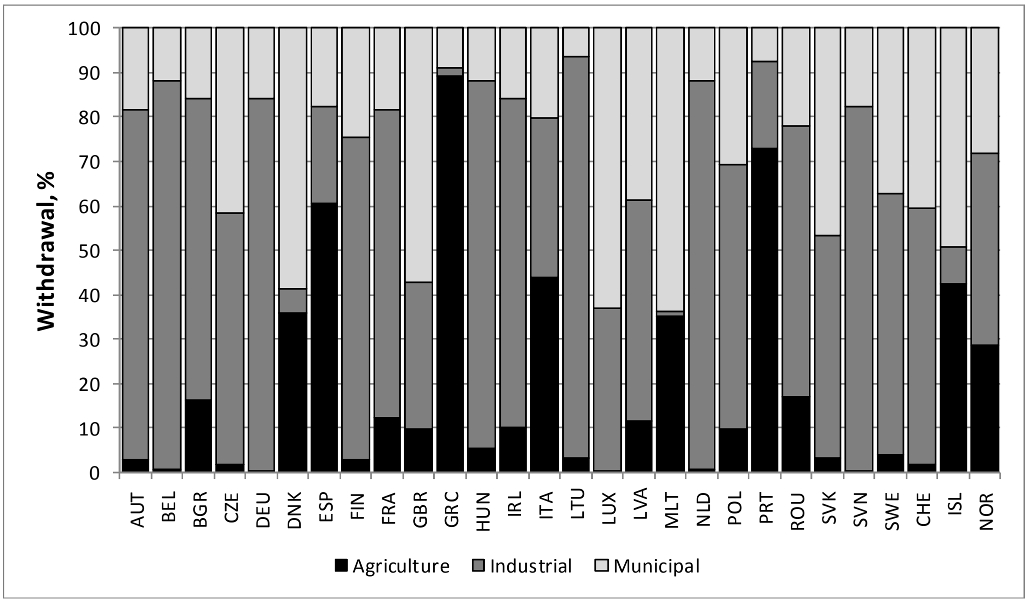

Figure 9: Proportion of water withdrawals by sector for European countries.

(Data from FAO/aquastat)

The data for municipal use standardised by population are compared with data from Eurostat water statistics for 2009 "Total freshwater abstraction by public water supply" (Figure 10). The agreement is generally good, with the exceptions of Malta (who's public water supply is largely drawn from seawater) and Poland, for which there is no obvious explanation. The median per capita withdrawal is 211 l/day (Eurostat) or 226 l/day

(FAO/Aquastat), with a few countries substantially greater. Thus, the total energy consumption and carbon footprints would vary widely, even if the energy intensity was similar.

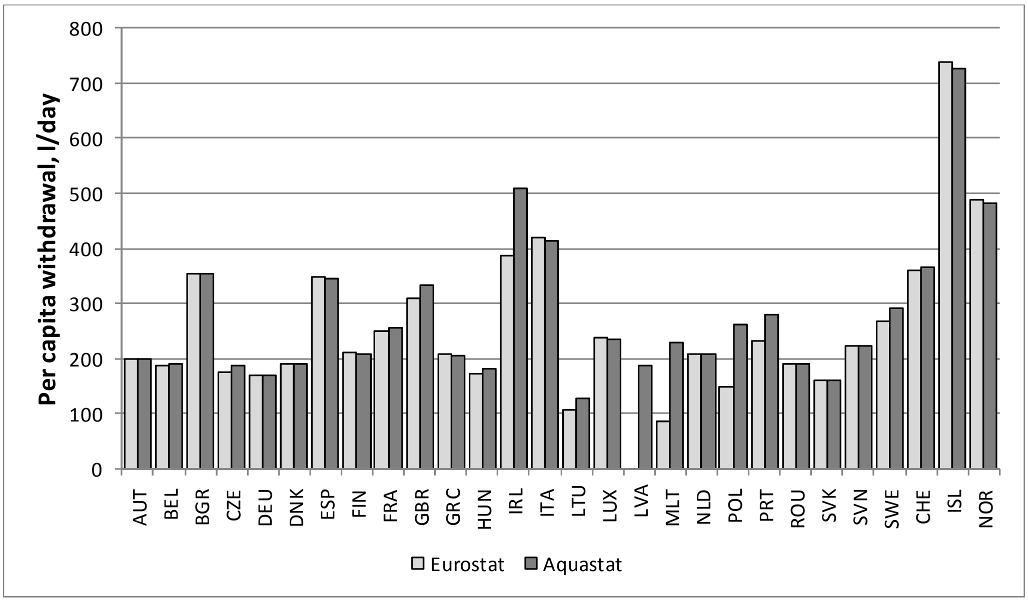

Figure 10: Per capita water withdrawals for municipal use by European countries.

(Data from FAO/aquastat and Eurostat)

The main water sources used also vary between countries. The use of desalinated sea water and reused treated water is currently a negligible proportion of the national totals throughout the EU (though desalination is locally important in some regions), so almost all water supplies come from fresh water: either groundwater or surface water. The proportions vary according to national circumstances from 98% groundwater in Denmark to over 90% surface water in several countries (Figure 11).

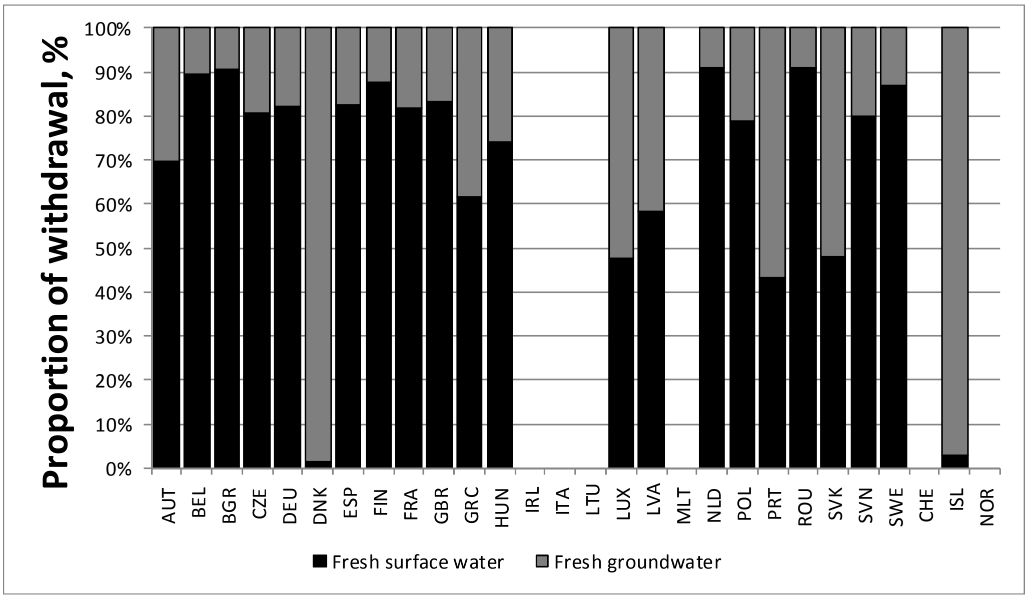

Figure 11: Proportions of surface water and groundwater withdrawals for European countries. (Data from FAO/aquastat)

3. ENERGY CONSUMPTION AND GHG EMISSIONS IN THE WATER INDUSTRY

3.1. National emissions

Countries that are party to the Climate Change Convention submit national GHG inventories to the secretariat of the United Nations Framework Convention on Climate Change. These inventories include wastewater treatment as a specific item, but not water supply, which is included within other categories. The average total emissions from wastewater treatment from the EU 27 countries for the period 2005–2009 were 22,700 kt CO_2e/year, or 0.46% of total emissions. The national emissions vary widely, in a similar pattern to municipal water use (Figure 12). It should be noted that these results include the direct emissions from wastewater treatment only: the emissions from the generation of electricity drawn from public utilities are accounted separately. The wastewater volumes from most countries are not readily available, but standardising by population shows that the emissions for most countries are 20–60 kg CO_2e person^{-1}year^{-1}, with about half under 40 kg CO_2e person^{-1} year^{-1} (Figure 13).

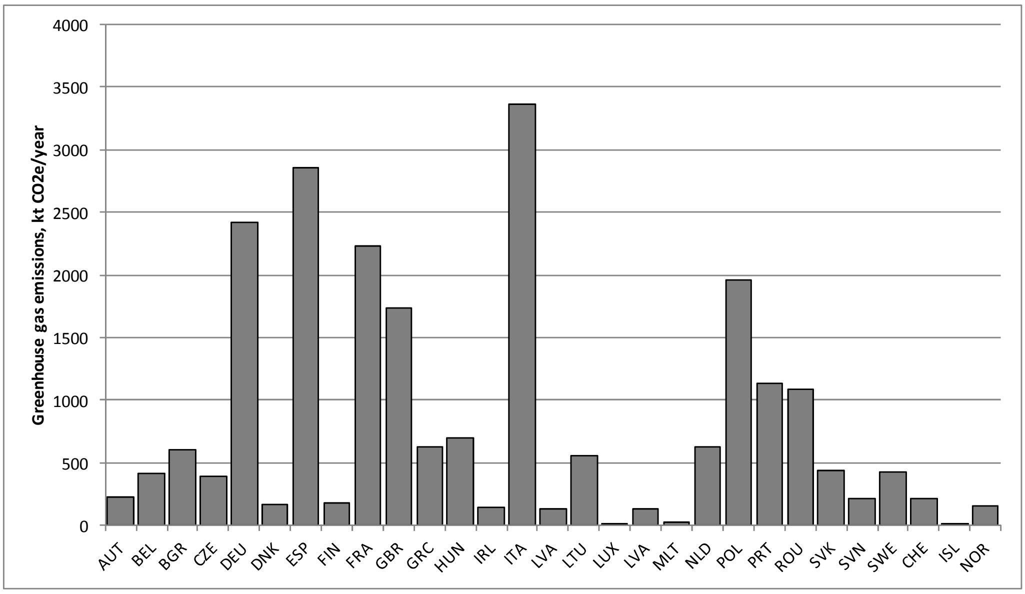

Figure 12: Average GHG emissions from domestic and commercial wastewater handling and treatment 2005-2009.

(Data from UNFCC http://unfccc.int/di/FlexibleQueries/)

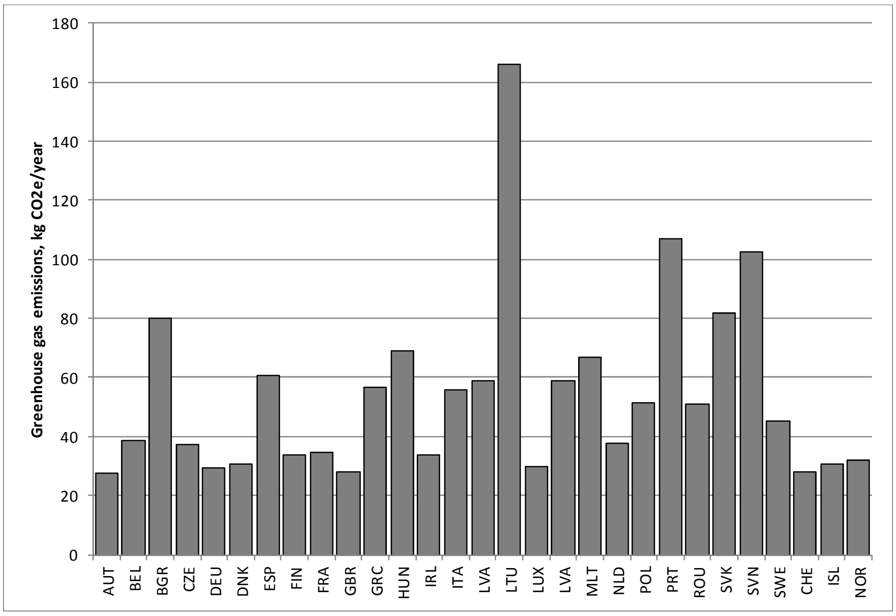

Figure 13: Average per capita GHG emissions from domestic and commercial wastewater handling and treatment 2005-2009 (Data from UNFCC http://unfccc.int/di/FlexibleQueries/)

Carbon footprint studies of the water sector also give estimates of the total emissions. Because the boundaries for the assessment are different from those used in the UNFCC inventory, typically including indirect emissions that are accounted elsewhere in the inventory, the resulting estimates for wastewater treatment are usually higher. They also include clean water treatment and supply, which are not visible in the inventory.

Relevant data are available from several countries within the EU. In the UK, the Climate Change Act 2008 imposed overall GHG reduction targets of 26% by 2020 and 80% by 2050 against a 1990 baseline. The CRC Energy Efficiency Scheme (introduced under the enabling powers of the Act) is a mandatory cap-and-trade scheme for all business that are not energy intensive, including the water sector (Georges *et al.*, 2009). As a result, data appear in water company annual reports and trade journals, as well as government publications and research papers.

The Defra Future Water Strategy mapped the emissions in 2005–06. Of the total footprint of about 5 Mt CO_2e/year, 56% were from wastewater treatment, 39% from clean water supply (all uses) and 5% from administration and transport (Defra, 2008). The total of 2.3 Mt CO_2e/year is significantly higher than the 1.7 Mt CO_2e/year in the inventory (Figure 12), presumably due to the inclusion of indirect emissions from electricity generation. Similar proportions were given by Anglian Water (UK) in 2011: 39% from water supply and 54% from wastewater and sludge treatment. For Scottish Water in 2008–09, the proportions were 25% for drinking water supply and 70% for wastewater and sludge treatment.

Table 1: UK water sector GHG emissions 2005–06 (from Defra, 2008)

SOURCE		GHG EMISSIONS, MT CO_2e/YEAR
Clean water supply and treatment to potable standard	[1].	1.0
Clean water distribution	[2].	0.6
Leakage	[3].	0.4
Wastewater pumping and collection	[4].	0.2
Wastewater treatment	[5].	2.1

A carbon footprint study of the water sector in the Netherlands reported a total footprint of 1.67 Mt CO_2e/year, of which 26% was from clean water supply and 74% was from sewerage and wastewater treatment (Frijns, 2012). Again this is higher than the 0.6 Mt CO_2e/year for wastewater in the inventory.

For comparison, a study in the USA, which considered energy use only, estimated the footprint of domestic and commercial water supply as 20.6 Mt CO_2e/year and that of wastewater treatment as 45.4 Mt CO_2e/year (Griffiths-Sattenspiel & Wilson, 2009). The corresponding UNFCC inventory figure for wastewater is 20 Mt CO_2e/year.

The pattern that emerges from this small selection of estimates is that the total direct and indirect emissions related to wastewater are substantially larger than those shown in the UNFCC inventory, and that emissions related to water supply are about one-third to two-thirds of those from wastewater. Consequently, to make reductions in GHG emissions associated with the water sector of the scale required by the UK Climate Change Act and the EU Climate Change Package, it is necessary to change both aspects. We will now consider the energy intensity and emissions of each in more detail.

3.2. Water supply

The three major components of energy consumption associated with water supply are extraction including conveyance to the treatment works, treatment to potable standard and distribution. These are not always reported separately.

Groundwater extraction by pumping is energy intensive: raising water by 1 m is estimated to require 3.5–7 Wh/m^3, with typical totals for the USA of 0.14–0.6 kWh/m^3 (Griffiths-Sattenspiel & Wilson, 2009). Conversely, gravity-fed surface water sources may require little or no energy for pumping. Similarly, the energy intensity of transporting water depends on the distances and lift heights involved. An extreme example is water delivered to Southern California from the Sacramento-San Joaquin Delta, over the Tehachapi Mountains, which requires 2.4 kWh/m^3. Unless the electricity used is supplied from renewable sources, its emissions will depend on the national supply for the country: lowest in countries with extensive geothermal or hydro-electric supplies (e.g. Sweden 0.023 kg CO_2e/kWh, Austria 0.209 kg CO_2e/kWh) or nuclear generation (e.g. France 0.056 kg CO_2e/kWh), but much higher in those dependent on coal (e.g. Poland 1.191 kg CO_2e/kWh), with intermediate values for countries using a mix including coal, oil, gas, nuclear and other sources (e.g. UK 0.55 kg CO_2e/kWh) (SEAP, 2010; AEA, 2011).

Many different treatments steps may be used (see Table 2: Treatment steps for potable water production (from Vince et al., 2008)), so the energy requirements vary widely, depending on the type and concentration of the contaminants to be removed. For example, filtration steps remove particles of various sizes, including bacteria (microfiltration and ultrafiltration) and viruses (nanofiltration). Others are used to remove various types of chemicals (e.g. GAC and reverse osmosis), or for sterilization (UV radiation).

Table 2: Treatment steps for potable water production (from Vince et al., 2008*)*

CLARIFICATION	FILTRATION	MEMBRANE TREATMENTS	DISINFECTION
Coagulation Flocculation Decantation Flotation	Sand filter GAC filter Dual filter	Prefiltration Microfiltration Ultrafiltration Nanofiltration Reverse osmosis	Ozonation UV radiation Oxidation
Chemical treatments	Thermal distillation	Other treatments	Water transfer
PAC injection Remineralization Neutralization	Multi-stage Flash Multi-effects MVC	Electrodialysis Biological treatment Ion exchange	Intake pumping Potable water distribution

An estimate for the UK in 2005–06 found that extraction and treatment combined emitted about 1 Mt CO_2e/year (0.178 kg CO_2e/m^3), and distribution (including leakage) a similar amount (Defra, 2008). A carbon footprint study (Reffold *et al.*, 2008) found UK average emissions of 0.03 kg CO_2e/m^3 for source, abstraction and conveyance, 0.14 kg CO_2e/m^3 for treatment and 0.11 kg CO_2e/m^3 for distribution. More recent data (Water UK, 2010) showed an average for source, treatment and distribution of 0.340 kg CO_2e/m^3 in 2010, varying between companies in the range 0.17–0.5 kg CO_2e/m^3 (Figure 14) for the reasons discussed above.

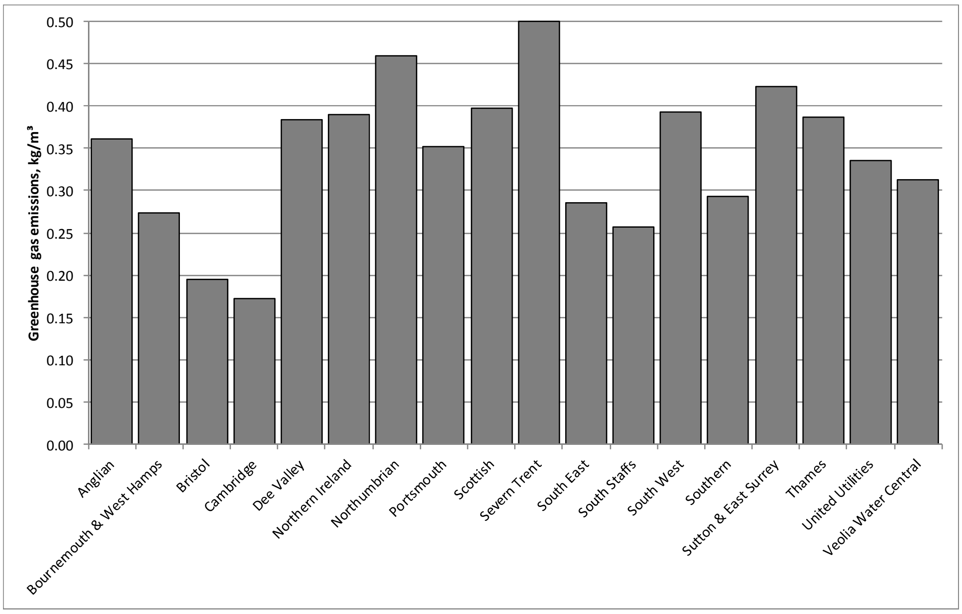

Figure 14: Greenhouse gas emission intensity of water treatment by UK water companies (source Water UK, 2010)

A study in the Netherlands found the average energy use for production and distribution of 789×10^6 m^3/year was 0.47 kWh/m^3, equivalent to 0.24 kg CO_2e/m^3 using an emission factor of 0.59 kg CO_2e/kWh (Frijns, 2012; Hofman *et al.*, 2009). In addition, there were direct emissions of carbon dioxide (10 kt CO_2e/year) and methane (37 kt CO_2e/year) from degassing and nitrous oxide (0.745 kt CO_2e/year) from ozonation (Frijns, 2012), totalling about 0.06 kg CO_2e/m^3. There has been an increase in energy consumption of 11% over the period 1997–2009 due to investment in central softening, which reduces household energy use, and new treatment steps, such as UV and membrane filtration (Frijns, 2012).

A detailed life cycle analysis (LCA) study of different treatment scenarios for both fresh and brackish water sources in France illustrated the range of energy intensities and GHG emissions that are possible (Vince *et al.*, 2008). Using freshwater, the energy required was 0.05–0.7 kWh/m^3, and intake pumping required a further 0.05–1.0 kWh/m^3. The range of consumption was further broken down by the type of treatment needed: conventional freshwater treatment 0.05–0.15 kWh/m^3, UF/MF membrane treatment 0.1–0.2 kWh/m^3 and advanced membrane treatment 0.4–0.7 kWh/m^3. Brackish water desalination by nanofiltration and reverse osmosis consumed 0.4–0.7 kWh/m^3 and saltwater desalination consumed 3.5–4.5 kWh/m^3 with energy recovery, or 5.5–7 kWh/m^3 without. Substantial GHG emissions also arose from the production of chemicals used in the treatment process, particularly lime (*c* 0.080 kg CO_2e/m^3) and ferrochloride (*c* 0.065 kg CO_2e/m^3). The total footprint for the ultrafiltration plant, including construction, was 0.289 kg CO_2e/m^3.

LCA was also applied to the Ebro River Water Transfer project, which formed part of the Spanish National Hydrologic Plan (SNHP), approved by parliament in July 2001 (Raluy *et al.*, 2005b). It was intended to divert 10^6 m^3/year between hydrological basins in Catalonia, Valencia, Murcia and Almería. Of this, 44% was intended for urban consumption and the rest was for irrigation and ecological restoration. The energy required for transfer was up to 4.067 kWh/m^3, depending on the destination. The study estimated the average emissions to be 1.44 kg CO_2/m^3 over 25 years or 1.55 kg CO_2/m^3 over 50 years. (The paper gives CO_2 rather than CO_2e, so it is unclear whether this is the total global warming potential.) As in the case of California, this illustrates the high energy and environmental costs of large-scale, long-distance transfers of water. The results for the relatively short northern path of the scheme, which included an urban supply to Barcelona were not reported separately, so cannot be compared with the assessments of other urban water supplies. In 2004 the new government abandoned the plan in favour of using desalination plants located close to the points of use.

The studies by Vince *et al*. and Raluy *et al*. are some of the few to include the impacts of construction and decommissioning; most others consider the operational burdens only. However, the results given by Vince *et al*. (2008) are based on other published analyses (e.g. Raluy *et al*., 2005a, Raluy *et al*., 2005b), rather than a direct analysis. In general, construction for a range of plant types, including conventional treatment and desalination was found to produce 5–15% of total emissions, while decommissioning was negligible (<1%). A range of 10–15% was found in an assessment of options for a new development in Melbourne (Sharma e*t al.,* 2009).

A comparison of the carbon footprints of tap water and bottled water in Siena found that the energy intensity of the tap water supply was 1.17 kWh/m^3 and the emissions were 0.9 kg CO_2e/m^3 (Botto *et al*., 2011). Of the total emissions, 94% were due to energy use, of which 97.5% was electricity. Construction and decommissioning were not included.

An energy efficiency study of two large municipal supplies in northern Portugal, supplying over 2,000,000 inhabitants, found average emissions of 0.25 kg CO_2e/m^3 from operational energy use (ManagEnergy, 2004). Using an average emission factor for Portugal of 0.369 kg CO_2e/kWh (SEAP, 2010) implies an energy intensity of 0.68 kWh/m^3.

A set of general estimates of energy intensity in Europe gave a range of 0.5–4 kWh/m^3 for surface water, 1–6 kWh/m^3 for recycled water, and 4–8 kWh/m^3 for desalination (WssTP, 2011). The results reviewed above are generally at the lower end of the range given for surface water, except when long-distance distribution is required.

We will now consider a sample of assessments from outside Europe. Several of these are from regions with constrained water resources, which may be indicative when considering the future in southern Europe.

A detailed carbon footprint analysis of different supply and sanitation options was carried out for the eThekwini Municipality of Durban in South Africa, a country with rainfall of only 500 mm/year (Friedrich *et al*., 2009). The objectives included provision of water to 63,000 additional households, with a target of 6 m^3/month for each household. The water supply options considered were the expansion of an existing dam and treatment works or the construction of new ones.

For the existing dam, the emissions related to construction averaged 8.8 g CO_2e/m^3 over 60 years (worst case) and the operational emissions were 0.051 kg CO_2e/m^3, giving a total of 0.06 kg CO_2e/m^3. The treatment works chosen for the study was the least efficient one serving the municipality. An earlier LCA study found the energy intensity was 0.1 kWh/m^3 and the emissions were 0.22 kg CO_2e/m^3. The energy intensity of distribution was 0.10 kWh/m^3 distributed or 0.13 kWh/m^3 delivered (taking leakage into account) with emissions of 0.14 kg CO_2e/m^3 distributed. Thus the total operational energy intensity and emissions were 0.2 kWh/m^3 and 0.41 kg CO_2e/m^3 distributed, or 0.53 kg CO_2e/m^3 delivered. Maximising the use of the existing assets could produce an additional 10^5 m^3/day; the effect on emissions would be a small reduction in emission intensity from the dam to 0.034 kg CO_2e/m^3.

Construction of a new dam and treatment works could produce three times the volume of water, with GHG emissions of 0.016 kg CO_2e/m^3. The new treatment works was assumed to use the same processes as the old one, but benefit from improvements in efficiency, reducing the footprint to 0.13 kg CO_2e/m^3. The energy required for distribution was unchanged, but it was assumed that leakage would be reduced to 20%. Thus the total footprint would be 0.29 kg CO_2e/m^3 distributed or 0.34 kg CO_2e/m^3 delivered.

A study of the water-energy relationship in California (Klein *et al.*, 2005) reported very wide ranges of energy intensity for water supplies in the USA. Extraction and conveyance ranged from 0 for gravity-fed surface water to 3.6 kWh/m^3. Treatment required 0.026–4.2 kWh/m^3 (the upper figure includes desalination) and distribution 0.066–0.32 kWh/m^3. The typical totals for northern and southern California were 0.83 and 2.7 kWh/m^3, the difference being entirely due extraction and conveyance from sources in the north.

These data are used in many other studies from the USA (e.g. Griffiths-Sattenspiel & Wilson, 2009). Griffiths-Sattenspiel & Wilson (2009) break down the extraction and conveyance options to include 0.53 kWh/m^3 for groundwater, 0.29 kWh/m^3 for recycled water, 0.84 kWh/m^3 for brackish water and 3.6 kWh/m^3 for seawater desalination. However, there is a risk of double-counting if these are combined with the values for treatment, which also include desalination.

A consultants' report for the California Public Utilities Commission (GEI Consultants & Navigant Consulting, 2009) also used the data from Klein *et al.* (2005), but included specific data for supplies in Los Angeles and San Diego. For Los Angeles, the energy intensity was 0–2.0 kWh/m^3 for conveyance, less than 0.035 kWh/m^3 for most treatment plants, but up to 0.42 kWh/m^3 for some groundwater wells, and 0.31 kWh/m^3 for distribution. For the major source of water to San Diego, the Colorado River, the energy for conveyance was 1.7 kWh/m^3 and treatment was only 0.041 kWh/m^3 at the largest plant. These results again emphasise the importance of conveyance when local water sources are inadequate to meet the demand.

In contrast to California, it is reported that the utilities in Wisconsin use 0.4–0.5 kWh/m^3 for treatment and distribution, including the effects of leakage,(SAIC, 2006). This is comparable to the values for Europe above.

Because of the difficulties of collecting the process data, an economic input-output LCA approach was used to consider the water supply to Kalamazoo, Michigan, in the Great Lakes region of the USA (Mo *et al.*, 2010). Economic input-output LCA uses standard factors to convert financial costs to environmental burdens and implicitly includes all direct and indirect inputs and outputs including construction, but does not give separate results for the three phases of supply. The estimated total energy intensity was 2.6 kWh/m^3 and the footprint was 1.7 kg CO_2e/m^3, primarily from coal-fired generation. These values appear high for a water-rich region, which may be a consequence of the inclusion of more indirect emissions.

A hybrid LCA approach, using economic input-output LCA supplemented by process data was applied to southern California (Stokes & Horvath, 2009). For the current system in which water is conveyed from the north of the state, the energy intensity was 5 kWh/m^3 and the GWP was 1.1 kg CO_2e/m^3. The results for recycled water were similar, and rose to 7.5 kWh/m^3 (1.6 kg CO_2e/m^3) for brackish water and 12 kWh/m^3 (2.4 kg CO_2e/m^3) for desalinated sea water. As with Mo *et al.*, these are higher than found by the other studies.

Economic input-output LCA was also used to analyse the impact of chemical manufacturing in the water supply system for Toronto, Canada, while process values were used for operational energy (Racoviceanu *et al.*, 2007). The energy intensity was 0.64–0.69 kWh/m^3 and the GWP was 0.13 kg CO_2e/m^3. The operational burdens, principally on-site pumping (60%) accounted for 94% of energy use and 90% of GHG emissions. The GHG emissions were relatively low because of the use of hydro-electricity in Toronto.

LCA was used to compare a hypothetical conventional plant enhanced with granular activated charcoal (GAC) and an existing nanofiltration plant in Quebec (Bonton *et al.*, 2012). The plants were designed to treat fresh surface water from a lake with a high organic matter content to the same standard. The footprints were 0.7 kg CO_2e/m^3 (90% 0.5–1.11) for the conventional-GAC plant and 0.05 kg CO_2e/m^3 (90% 0.045–0.057) for nanofiltration, including construction. The nanofiltration plant had a much higher energy intensity (0.55 kWh/m^3 compared with 0.16), but the main source in Quebec is hydro-electricity, so its effect on the footprint was small. Using the energy mix for the USA the footprints were 0.8 kg CO_2e/m^3 (90% 0.6–1.2) for the conventional-GAC plant and 0.48 kg CO_2e/m^3 (90% 0.4–0.7) for nanofiltration.

Operational power consumption for water treatment and supply in four Australian cities (Sydney, Perth, Melbourne and Brisbane) was 0.13–0.46 kWh/m^3, of which treatment was 0.02–0.15 kWh/m^3 and pumping 0.11–0.31 kWh/m^3 (GWRC, 2008).

These results are summarised in Table 3. In general the energy intensity is in the range 0.2–1.0 kWh/m^3 except where there are very large pumping requirements, such as southern California. The exceptions are the two studies using EOI LCA, which may include a larger component of indirect energy use. Most carbon footprints are in the range 0.1–0.9 kg CO_2e/m^3; the variation in energy sources introduces an additional source of variation between the case studies.

One omission from most of the studies other than Bonton *et al.* is an uncertainty estimate. The usual practice, recommended by the IPCC, is to include uncertainties (e.g. standard deviations) for all of the input variables and use Monte-Carlo simulation to estimate a mean and 95% confidence interval for the resulting emissions.

Table 3: Energy intensity and GHG emissions from water supply

COUNTRY	ENERGY INTENSITY, kWh/m^3	GHG EMISSIONS, kg CO_2e/ m^3	SOURCE	NOTES
UK		0.34 (0.17–0.5)	Water UK, 2010	Extraction, treatment and distribution variation between companies)
Netherlands	0.47	0.24 (0.06)	Frijns, 2012	Production and distribution (additional direct emissions)
France		0.289	Vince et al., 2008	Intake pumping and ultrafiltration

Spain		1.44	Raluy et al., 2005b	Includes long-distance transfer
Italy	1.17	0.9	Botto et al., 2011	Operation, including maintenance
Portugal	0.68	0.25	ManagEnergy, 2004	Operational energy use
Europe	0.5–4		WssTP, 2011	Surface water systems
South Africa	0.26	0.53 (0.34)	Friedrich et al., 2009	Construction and operation using existing (new) dam and plant
USA	0.8 2.7		Klein et al., 2005	Northern CA Southern CA
USA	0.35–0.73 1.74		GEI Consultants & Navigant Consulting, 2009	LA (exc. conveyance) SD (incl. conveyance)
USA	0.4–0.5		SAIC, 2006	Wisconsin
USA	2.6[1]	1.7	Mo et al., 2010	Michigan, EIO LCA
USA	5	1.1	Stokes & Horvath, 2009	Southern CA, EIO LCA
Canada	0.64–0.69	0.13	Racoviceanu et al., 2007)	Using hydro power
Canada		0.7 (0.8) 0.05 (0.48)	Bonton et al., 2012	GAC NF Hydro (thermal) generation Incl. construction.
Australia	0.13–0.46		GWRC, 2008	

[1] Based on a calculation of primary energy

3.3. Wastewater treatment

The proportion of the population connected to wastewater treatment systems, and hence the volume treated, varies widely across Europe (Figure 15), reflecting urbanisation, economic development and other factors (Eurostat, 2010).

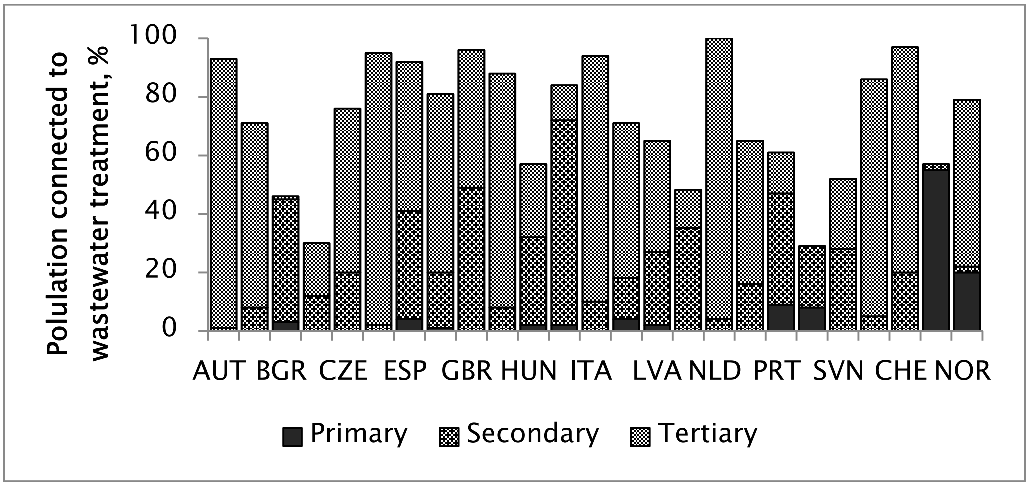

Figure 15: Proportion of the population connected to wastewater treatment in European countries (source: Eurostat)

The stages in wastewater treatment were outlined in Section 2.1. The treatments applied and their efficiency will depend on the nature of the waste, the design of the treatment works and its size. As with water supply, electricity is required for pumping of sewerage, within the treatment plant and for discharge.

> "In secondary treatment, most of the energy is used for biological treatment; pumping of wastewater, liquid sludge, biosolids and process water; and processing, dewatering, and drying of solids and biosolids. Tertiary treatment requires additional energy for aeration, pumping, and solids processing." (Klein *et al.*, 2005)

Sedimentation uses chemicals with related life cycle emissions. A large proportion of wastewater treatment works across Europe utilise the activated sludge process as the main secondary treatment stage. The energy requirements to operate this process are fairly high: 0.15–0.7 kWh/m^3 (WssTP, 2011). Other aerobic secondary treatments for the liquid effluent are also be energy intensive because of the use of electricity for aeration. Sludge dewatering and drying both require energy, whereas digestion produces biogas, so can be energy positive if this is utilised (which also reduces methane emissions). Using dried sludge as a fuel also produces energy, but incineration may not. Tertiary treatments, including filtration and chemical of ultraviolet disinfection may be applied to the clear effluent before discharge or reuse.

Sludge applied to farmland will continue to emit greenhouse gases including nitrous oxide, which is also emitted during treatment. The variation in the amount of sludge produced largely reflects the size of the population connected to wastewater treatment (Figure 16). The end use and disposal options vary between countries, from 80% agricultural use in Spain and Cyprus, to 100% landfill in Greece, Malta and Iceland, and 100% incineration in the Netherlands, where spreading of all wastes to agricultural land is tightly regulated (Figure 17). The reduction of biodegradable waste disposed of in landfill is now one of the principal aims of the EU Landfill Directive (1999/31/EC), so this pattern is likely to change in the future.

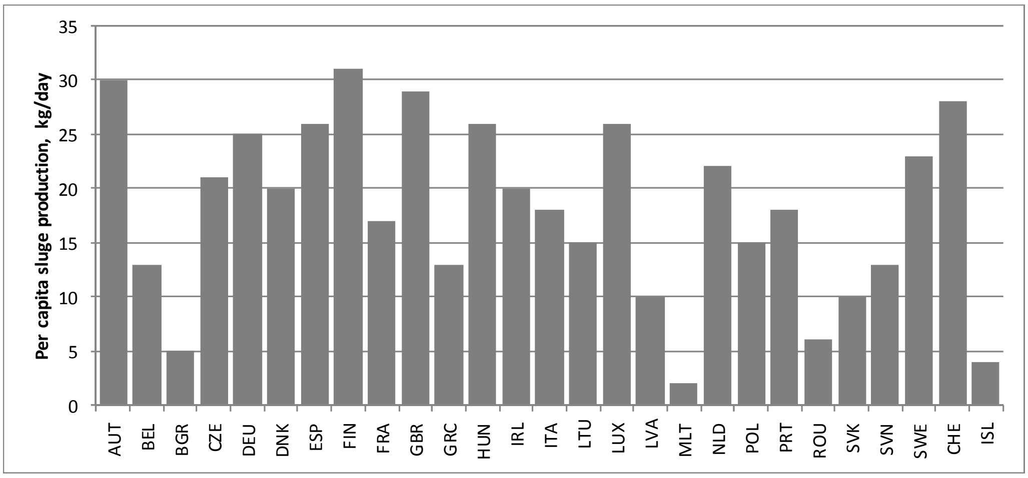

Figure 16: Per capital production of sewage sludge in European countries (source: Eurostat)

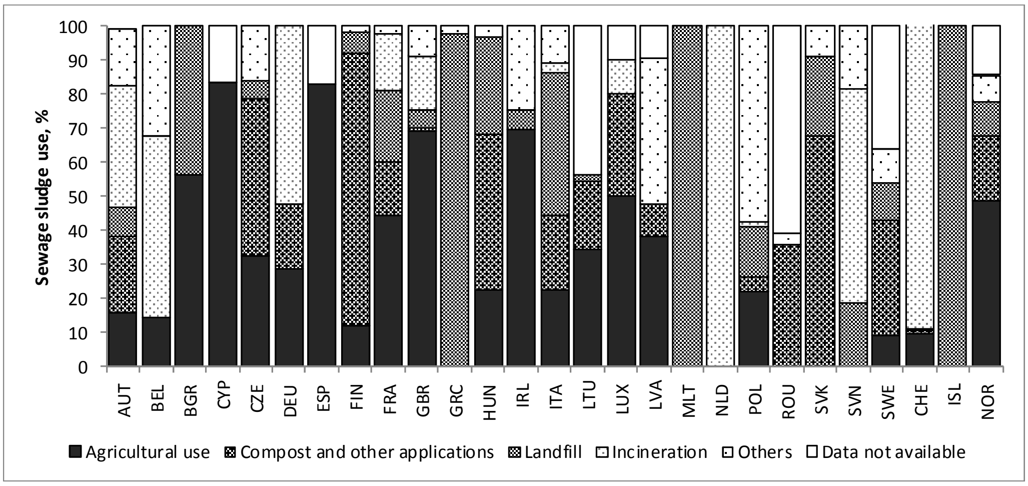

Figure 17: Use and disposal of sewage sludge in European countries (source: Eurostat)

With current treatment plants the process is energy intensive and produces significant direct emissions of methane and nitrous oxide. As noted in Section 3.1, the GHG emissions from wastewater treatment are typically 1.5–3 times those from water supply, although the processed is less than the water supplied. For example, Anglian Water in the UK supplies 1.2×10^6 m^3/day and treats 0.9×10^6 m^3/day of wastewater, including some areas for which it is not the water supplier, but the associated emissions are 542 t CO_2e/day for water supply and 741 t CO_2e/day for wastewater treatment (Anglian Water, 2011b;Anglian Water, 2011a). For the UK as a whole, the footprint of wastewater treatment is 2.1 Mt CO_2e/year, with a further 1–2 Mt CO_2e/year due to land application of sludge. The average emissions intensity in 2010 was 0.7 kg CO_2e/m^3, varying between companies in the range 0.4–1.1 kg CO_2e/m^3 (Figure 18; Water UK, 2010).

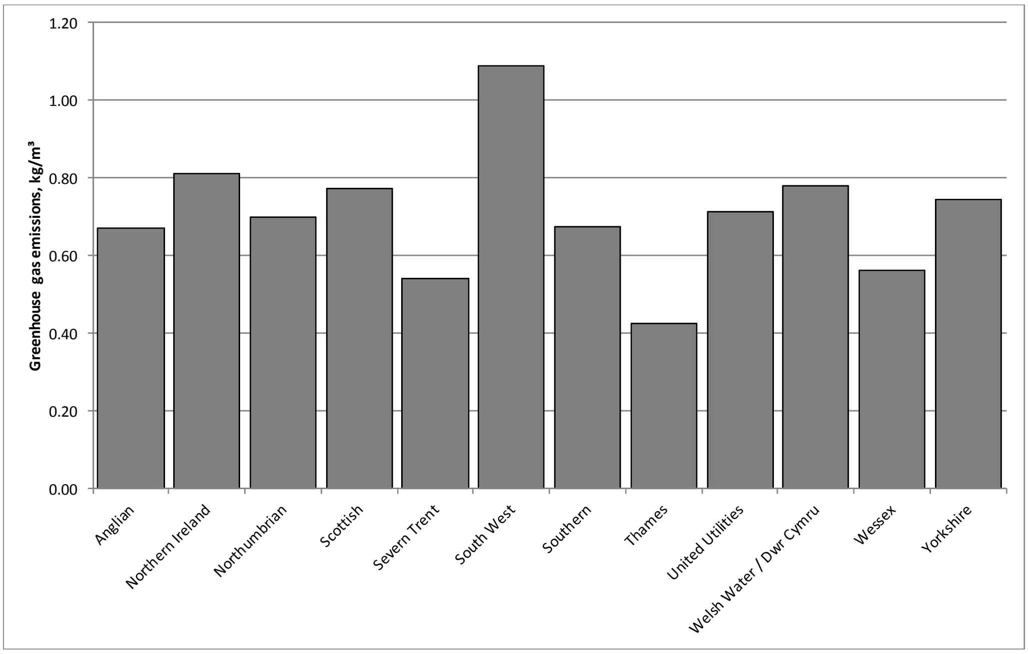

Figure 18: Greenhouse gas emission intensity of wastewater treatment by UK companies (source Water UK, 2010)

Considering UK domestic use only, Reffold *et al.* (2008) used an average of 0.476 kg CO_2e/m^3 water supplied for wastewater treatment. From the other values given, pumping produced 0.187 kg CO_2e/m^3 and treatment 0.289 kg CO_2e/m^3. However, some of the supplied water would enter the wastewater system, so the intensities for wastewater would be somewhat higher.

An international workshop in 2008 summarised the energy intensity of wastewater treatment in several countries, including The Netherlands 0.36 kWh/m^3, Germany (including collection) 0.67 kWh/m^3, USA 0.45 kWh/m^3 and Australia 0.39 kWh/m^3 (GWRC, 2008).

Frijns (2012) estimated that wastewater produced 67% (1.1 Mt CO_2e/year) of the carbon footprint of the water sector in the Netherlands, and sewerage a further 8% (0.13 Mt CO_2e/year). Of this total, direct emissions of methane and nitrous oxide from wastewater were 0.56 Mt CO_2e/year. The estimate for sewerage was based on an energy intensity estimate of 0.11 kWh/m^3 from limited practical data. The total volume entering wastewater treatment was 3.3×10^9 m^3/year, of which 38% was from non-domestic sources including rainwater, groundwater and industry (Hofman *et al.*, 2009). The energy input was about 1.5×10^9 kWh/year, so the energy intensity for treatment was 0.46 kWh/m^3. Based on these volumes, the GHG emission intensity was about 0.33 kg CO_2e/m^3 treated. These values are comparable to those for the UK and in good agreement with that reported by GWRC (2008).

Analysis of data from 177 water treatment works in Valencia treating 146,000 m^3/year found that the mean energy intensity was 0.821 kWh/m^3 (standard deviation 0.532), compared with the national average of 0.53 kWh/m^3 (Hernández-Sancho *et al.*, 2011). The efficiency varied with the size of the plant, from 0.963 kWh/m^3 for those treating less than 100,000 m^3/year to 0.486 kWh/m^3 for those over 250,000 m^3/year.

The carbon footprint analysis of different supply and sanitation options for the eThekwini Municipality of Durban in South Africa also considered wastewater treatment options (Friedrich *et al.*, 2009). They found emissions of 0.15 kg CO_2e/m^3 from collection of sewage, 0.112 kg CO_2e/m^3 from primary treatment and 0.297 kg CO_2e/m^3 from secondary treatment, of which the main component was the consumption of electricity, with the activated sludge process being the major consumer. They also considered tertiary treatment in a recycling plant, including chemical treatment, settling, filtration, ozonation and GAC, which resulted in emissions of 0.94 kg CO_2e/m^3.

In California, Klein *et al.* (2005) found that the average energy required to transport sewage from customer to treatment, taking advantage of gravity feeds where possible, was 0.04 kWh/m^3. The energy intensities of seven treatment works were 0.505–0.785 kWh/m^3. The average for southern California was 0.53 kWh/m^3 water supplied (Means, 2003). A wider range of intensities was found in the USA when considering different plant types and sizes, from 0.18 kWh/m^3 for a large plant using a basic trickling filter to 0.78 kWh/m^3 to a small, advanced plant with nitrification (Slaa, 2011). Extreme cases may reach 1.6 kWh/m^3 (Griffiths-Sattenspiel & Wilson, 2009). A single township (470 m^3/day) treatment plant in the Pennsylvania (Ambulkar *et al.*, 2011) had an energy intensity of 0.83 kWh/m^3, and its GHG emissions, excluding disposal, were 0.67 kg CO_2e/m^3.

A Canadian study considered the effect on several environmental indicators of the control strategies applied to a model of a benchmark treatment works using the activated sludge process and anaerobic digestion, including disposal of the sludge (Flores-Alsina *et al.*, 2011). They found that the range of emissions was 0.87–1.14 kg CO_2e/m^3. Of these, about half were direct emissions from secondary treatment, a quarter were from sludge processing and most of the remainder from disposal. The use of power was largely offset by generation from biogas. The control strategies with the lowest emissions tended to result in high effluent concentrations, which would probably be unacceptable. These are at the upper end of the range reported in the other studies, but there is insufficient detail in most to assess whether this is due to influent conditions, plant design or the choice of boundaries.

The results are summarised in Table 4. The energy intensity found in most cases was 0.36–0.96 kWh/m^3 with some basic plants as low as 0.18 kWh/m^3 wastewater and extremes of up to 1.6 kWh/m^3. GHG emission intensities were less often reported, but similarly variable: 0.33–1.14 kg CO_2e/m^3, with one case where tertiary treatment added 0.94 kg CO_2e/m^3.

Because of its potency as a greenhouse gas (GWP 298 times CO_2), emissions of nitrous oxide from wastewater treatment have been studied in some detail (e.g. Foley *et al.*, 2010; Kampschreur *et al.*, 2009; Tallec *et al.*, 2006). The emission factors are highly variable, for example a review of 11 published studies on nitrous oxide emissions from full-scale and experimental systems found a median emission factor of 0.01 kg N2O-N/kg N influent with a range of 0.0003–0.03 (Foley & Lant, 2008). The emissions vary as a result of influent concentration, dissolved oxygen levels, type of treatment and many other factors. Furthermore, nitrous oxide emissions may continue due to nitrification and denitrification after discharge. In view of the large uncertainty, no further quantification will be attempted here.

Table 4: Energy intensity and GHG emissions from wastewater treatment

COUNTRY	ENERGY INTENSITY, kWh/m^3	GHG EMISSIONS, kg CO_2e/m^3	SOURCE	NOTES
UK		0.7	Water UK, 2010	mean(variation between companies) /m^3 wastewater
UK		0.476	Reffold et al., 2008	Total pumping treatment /m^3 water
Netherlands	Frijns, 2012 /m^3 wastewater	0.289	Vince et al., 2008	Intake pumping and ultrafiltration
Netherlands	0.36		GWRC, 2008	/m^3 wastewater
Germany	0.67		GWRC, 2008 including collection	/m^3 wastewater

Spain	0.821 (0.486–0.963)		Hernández-Sancho et al., 2011	mean (range) /m^3 wastewater
Europe	0.15–0.7		WssTP, 2011	Activated sludge process only
South Africa		0.15 0.112 0.297 0.94	Friedrich et al., 2009	transport primary treatment secondary tertiary /m^3 wastewater
USA	0.505–0.785		Klein et al., 2005	California /m^3 wastewater
USA	0.53		Means, 2003	S. California /m^3 water)
USA	0.18–0.78		Slaa, 2011	m^3 wastewater
USA	up to 1.6		Griffiths- Sattenspiel & Wilson, 2009	m^3 wastewater
USA	0.83	0.67	Ambulkar et al., 2011	Pennsylvania /m^3 wastewater
USA	0.45		GWRC, 2008	m^3 wastewater
Canada		0.87–1.14	Flores-Alsina et al., 2011	Varying control strategy /m^3 wastewater
Australia	0.39		GWRC, 2008	/m^3 wastewater

4. ENERGY AND GHG IMPLICATIONS OF TREATMENT TECHNOLOGIES

Collating a reliable picture of energy use and GHG emissions for the various elements of the urban water cycle presents particular challenges. Variations in local circumstances, implemented technologies, asset age, operating regimes, etc. etc. make reliable projections across technology sets or geographical regions alarmingly unreliable. Consequently, allocating energy / GHG reduction potentials to anything other than a single system (abstraction – treatment – supply - treatment – discharge) or at a highly aggregated scale, similarly problematic. Table 5 (below) utilises data from a recent report to provide an indication of where energy is used in the urban water cycle and where savings potentials are located. The remainder of this section goes on to discuss energy and GHG reduction possibilities across a range of technology options.

Table 5: Distribution of energy consumption across the urban water cycle and indicatory savings potentials (Compiled by the authors from data listed in ESMAP, 2012)

ENERGY USING ACTIVITY	INDICATIVE ENERGY SHARE	ENERGY SAVING POTENTIAL	COMMENTS
Water Supply			
Raw Water Extraction (Pumping,	Surface Water: 10% Ground Water: 30%	5-10% by improving existing pumps. Up to 30% by better maintenance and closer matching to load 15% savings possible in buildings services	
Treatment (Mixing, Other treatment processes , Pumping (for backwash, etc.), Water sludge processing and disposal, Building services)	Surface Water: 10% Ground Water: 1%	Up to 20% but potable treatment already reasonably energy efficient	

Clean Water Transmission and Distribution (Pumping)	Surface Water: 80% Ground Water: 69%	5-10% by improving existing pumps. Up to 30% by better maintenance and closer matching to load	Dependent on the share of gravity—fed water supply
Wastewater Management (Assuming Activated Sludge Based Treatment)			
Wastewater Collection (Pumping	10%	5-10% by improving existing pumps. Up to 30% by better maintenance and closer matching to load	Dependent on the share of gravity-induced collection
Treatment (Aeration, Other treatment processes, Building services)	55%	20-50% through better alignment of control parameters with the discharge standard	Mostly for aeration of wastewater
Sludge Treatment and Disposal (Centrifugal and press dewatering, Sludge pumping, storing, and residue burial, Building service)	35%		Energy can be produced in sludge processing

4.1. Demand management

Most of the operational GHG emissions for water supply are directly proportional to the volume used, so demand reduction is an obvious approach to reducing emissions. However, the effect of reduced flows on wastewater treatment will be less, as the requirements are related to the chemical and biological loads, not to the volume of water (Stephenson, T. in

Ainger *et al.*, 2009), with the possible exception of dewatering processes. If, for example, collected rainwater were used for toilet flushing, the reduction in sewerage volumes would be less than that in water treatment, though there would be a corresponding reduction in urban drainage flows.

The differences in per capita extraction for municipal use between countries were noted in Section 2.3. This has raised the prospect of reducing the high consumers towards the lower ones. However, a fuller analysis of the differences would be required to evaluate how realistic this is.

Domestic consumption is only part of the total municipal consumption. For example, average consumption in the UK is usually quoted as 150 l/day, which is similar to Luxembourg, France, Finland and Hungary (Defra, 2008). However, for the UK the total per capita municipal withdrawal is about 300 l/day, whereas for the other four countries it is about 200 l/day. The domestic consumption data are taken from Waterwise (http://www.waterwise.org.uk/) and few details are available, so it is not clear whether the differences are real or artefacts of the methods.

The UK has target to reduce demand to 120–130 l/day by 2030, a reduction of about 20%. This is the current level in Germany, The Netherlands and Denmark (Defra, 2008), which have similar municipal withdrawals to Luxembourg, France, Finland and Hungary. It has been estimated (Reffold *et al.*, 2008) that the introduction of universal metering would reduce domestic demand by about 10% (or 5% of total municipal supply)

The Code for Sustainable Homes (DCLG, 2006) has set targets to reduce water use in new homes in England and Wales. The maximum predicted consumption to qualify is 120 l $person^{-1}day^{-1}$, and the highest category requires 80 l $person^{-1}day^{-1}$. Without a reduction in other aspects of water use, achieving an average of 120 l/day across all homes would amount to a reduction of about 10% of municipal consumption. Achieving larger reductions is likely to be challenging.

Hypothetically, if all the countries above the median per capita withdrawal in Figure 10 were able to reduce half way towards the median, the total municipal water use would be reduced by about $6×10^9$ m^3/year or 12%. Using the emission factor of 0.24 kg CO_2e/m^3 from Frijns (2012) which is at the lower end of the range reported (Table 3), this could reduce emissions by 1.4 Mt CO_2e/year.

Systems to reuse ‘grey’ water within homes for toilet flushing, garden use and washing machines are sometimes proposed as a method of reducing the demand for mains water. The public acceptability of such systems is mixed: in general, users prefer to use their own water to other people’s, and acceptability is greater where there is minimal contact, for example toilet flushing (Jeffrey, 2002). However, a survey of rainwater harvesting and grey water recycling schemes in the UK concluded that “Buildings using harvested rainwater or treated grey water typically increase greenhouse gas emissions compared to using mains water, where total cradle to gate embodied and operational carbon are considered” (Parkes *et al.*, 2010). It found that operational energy and GHG emission intensities rainwater of harvesting systems were 40% higher and most grey water recycling systems 100% higher than mains water. The exception was short residence time grey water recycling systems, which were about 40% less intensive. The energy use arises from the need to treat the water if it is being stored and to pump it within the home. Nevertheless, it may be necessary to install grey water systems to meet the higher tiers of the England and Wales Code for Sustainable Homes (http://www.bre.co.uk/page.jsp?id=47). Further research may reduce the carbon footprint of such systems (Parkes *et al.*, 2010).

Demand reduction in general requires behavioural changes by consumers to avoid waste and increases in the efficiency of appliances. Most of these are beyond the control of the industry, although many companies promote public awareness of the need to conserve water. It will not, therefore, be considered in more detail.

4.2. Dual supplies

In most countries, all municipal water supplies are of potable quality. Some uses, such as those suitable for grey water, could be supplied with water treated to a lower standard. In the UK, 26% of domestic supplies are used for toilet flushing, 12% for clothes washing and 7% for outdoor use, such as car washing and gardens (Environment Agency, 2011). In California about half of domestic supplies are for outdoor use (Griffiths-Sattenspiel & Wilson, 2009). It is unlikely that the usage is as high as this in most European countries, but it is still significant. Rainwater diversion can supply part of the need, but even in the UK it is least abundant at the time of highest demand.

Providing a separate supplies of potable water and water treated to a lower standard has been proposed as a method of reducing total energy use and GHG emissions. Several schemes supply recycled water for non-potable use via dual pipe systems in the USA (ACWA, 2011) and Australia (see http://www.recycledwater.com.au/) . Such a system for the three uses identified above for the UK would allow 45% to be treated to the lower standard. Data for non-domestic municipal use are more uncertain, but many of the same uses apply to civic and commercial buildings and to amenity use of water. However, in addition to the question of public acceptability, the implications are considerable. Dual distribution systems would be required from the water treatment works to the point of use, including rigorous separation at the end point. This is unlikely to be feasible except for new developments. The embodied carbon in the distribution system would be doubled. None of the LCA studies surveyed gave sufficient detail to identify this component, but the survey by Vince et al. (2008) found total embedded carbon in the water supply to be 5–15%, so this component is likely to be relatively small.

More significantly, only the treatment component would be reduced: extraction and distribution would be unchanged. In the studies considered above, distribution typically accounted for about half of the energy use, while the energy used for extraction was comparatively small except where long-distance conveyance was required. Assuming that 60% of the energy use and GHG emissions are associated with treatment, that the 45% non-potable estimate can be applied to total municipal water, and (arbitrarily) that the lower standard would require 30% less energy for treatment, the net reduction would be only 8%. Dual supplies will be considered further when discussing recycling in Section 4.8.

4.3. Conveyance and distribution

It is clear that long-distance and high-lift conveyance and distribution of water are energy intensive, so that using local supplies will normally be lower-energy options, except where low source quality entails intensive treatment. Other than embedded carbon in the infrastructure, the emissions associated with conveyance and distribution arise from electrical energy use for pumping. Distribution typically accounts for half of the energy use for the water supply. The main options for reducing the emissions are sustainable generation and efficiency improvements.

Proposed efficiency improvements include "pump optimisation, low friction linings, online energy/resource optimisation systems and pressure reduction systems, and smart and self-healing pipes" (Caffoor, I. in Ainger *et al.*, 2009; also Rothausen & Conway, 2011). It has been shown that implementing a range of such measures can reduce energy costs for pumping by 30–50% (Yates & Weybourne, 2001). However, some projects have achieved part of the saving by managing the peak load to reduce the unit price of electricity (ManagEnergy, 2004). Yates & Weybourne gave two case studies in which energy reductions of 16% and 18% could be achieved with relatively simple modifications and payback periods of 2–3 years. A survey of other case studies (Middleton & Frijns, 2010) found a range of savings including 37% from the use of variable speed drives, 19% from intrinsic pump system efficiency, 11% from duty point selection or variable duty selection, and 5–20% from change of duty. They estimated that pipework design could also save 5–20%.

Some of these improvements are only likely to be made in new installations or when components are due for replacement. However, when combined, reductions in operational energy consumption in the range suggested by Yates & Weybourne, hence 15–25% of the total for water supply, might be feasible in the medium to long term.

As pumping depends on electricity, its emissions depend crucially on the national generation mix. Thus it will be a lower proportion of the total where hydro generation or nuclear power is the dominant source. As other countries reduce their emissions from generation, either by moving away from fossil fuels, or by introducing carbon capture and storage, there will be a proportional reduction in emissions related to pumping. This is beyond the control of the water industry, but companies may be able to make use of renewable energy sources, which are considered below.

4.4. Water treatment

Most of the studies surveyed in Section 3.2 simply considered water treatment as a unit, reporting energy use by existing plants or national averages. A few considered treatment steps in greater detail, which enables 'hot spots' to be identified. GAC-filtration (including GAC production) and ozonation (including ozone production) are often regarded as energy intensive treatments in common use. In general, companies would be expected to avoid using energy-intensive treatments unnecessarily, because of the cost. However, water treatment works have long lifetimes, so they may have to use the existing treatment options where less energy-intensive ones exist but are not installed.

In the comparative LCA of a conventional plant with GAC and a nanofiltration plant (Bonton *et al.*, 2012), some data are given for operational consumption of electricity, although the categories do not correspond directly to individual processes. The conventional plant consisted of coagulation, flocculation and settling, granular filtration and GAC adsorption. The NF plant used pre-filtration then nanofiltration. Chlorination and corrosion control were common to both.

The largest energy user by a factor of 5 was pumping in the NF system (0.49 kWh/m^3). The only other operations above 1% of the total were pre-filtration (0.035 kWh/m^3) and lighting (0.025 kWh/m^3). For the conventional plant, the largest consumer was building heating (0.09 kWh/m^3), then mixing (0.035 kWh/m^3), pumping (0.029 kWh/m^3), lighting (0.006 kWh/m^3) and cleaning (0.0024 kWh/m^3).

Operational energy use does not give the complete picture for the conventional-GAC system, however. For this system, 88% of the GHG emissions were associated with replacement of the GAC, which demonstrates the importance of assessing the complete life cycle. Similarly, for the NF system, the GHG emissions associated with the chemicals used were about double those from electricity. Construction was negligible compared with the operational emissions.

As noted above, the baseline assessment assumed that most of the electricity came from hydro generation, so the results are slightly atypical. Similar results were found with the French generation mix, which has a high proportion of nuclear power. Using the emission factor for the USA, which is heavily dependent on coal, GAC replacement was still the dominant component for the conventional-GAC plant (I estimate 77%), but the emissions from the NF plant related to pumping increased by a factor of 8, to become the largest component.

The implications for Europe are clear: for energy-intensive operations the source of the energy is crucial in determining emissions; conversely, where low emission sources are available, other parts of the life cycle will dominate.

A more detailed breakdown of the impact of individual processes was given by a study that applied LCA to the expansion of Amsterdam's water supply from the River Rhine (Mohapatra *et al.*, 2002). The authors were interested in the total environmental impact, so all the results are given as 'Eco-points' using the Eco-indicator 95 system, which is an impact measure based on a weighted combination of the burdens in the life cycle inventory (Goedkoop *et al.*, 1996).

The existing supply (70 M m^3/year) uses dune infiltration followed by ozonation and GAC filtration. Dune infiltration with heavily polluted water is no longer ecologically acceptable, so the proposed expansion (13 M m^3/year) would use sand filtration and reverse osmosis, possibly with an initial ozonation step. The highest impacts in the current system came from several stages of pumping, softening and the GAC filtration (replacement of the GAC). In the alternative system, pumping had a higher impact than any of the individual stages in the conventional system, but lower than all the pumping steps combined. The impact of ozonation was comparatively small. Although it is not explicit in the paper, GHG emissions would be a significant component of the impact of pumping, through electricity generation.

A carbon footprint analysis of Amsterdam's water system, including both water supply and wastewater (Janse & Wiers, 2007) found that direct (Scope 1) emissions were only 20% of the total. These were equally divided between heating and nitrous oxide emissions. Scope 2 emissions from electricity consumption were 27%. The remainder were Scope 3 (indirect) emission associated with the use of energy, goods and materials. Chemical manufacturing, particularly sodium hydroxide and ferric chloride, was about 70% of this, and iron and steel a further 17%.

Frijns (2012) included assessments of some of the emissions from water treatment that are not related to energy use. Groundwater abstraction sites released both carbon-dioxide (10 kt/year) and methane (37 kt CO_2e/year), while ozonation released nitrous oxide (0.75 kt/year). These amounted to 8% of the total emissions from treatment. Frijns found that the embodied energy of the chemicals used in water treatment was typically responsible for only 3% of the total GWP. He notes that this may be higher for some surface water abstraction sites that require large amounts of sodium hydroxide and ferrous sulphate.

Vince *et al.* (2008) give both the direct energy inputs and the GHG emissions associated with a treatment plant for freshwater with a high organic matter content. The processes with the highest consumption were ultrafiltration (0.046 kWh/m^3), decantation (0.026 kWh/m^3), prefiltration (0.023 kWh/m^3) and ozone production (stated but not quantified). The processes with the largest GHG emissions were GAC (including production), ultrafiltration, ozonation (ozone production) and decantation. Each of these emitted about 20–30 g CO_2e/m^3. However, three other chemical production processes had much greater emissions: lime (80 g CO_2e/m^3) and carbon-dioxide (40 g CO_2e/m^3) for remineralisation, and ferrochloride (70 g CO_2e/m^3) for clarification. The UCTE electricity generation mix was assumed, so these results are not strongly influenced by French use of nuclear power.

Overall, the largest on-site user of energy in most cases was pumping, particularly in association with operations such as nanofiltration and ultrafiltration. For countries where electricity generation relies on fossil fuels, this was a significant component of the GHG emissions. Where the processes used are determined by the treatment requirement of the source, it is unlikely that big changes can be made, especially as asset lifetimes are usually measured in decades.

4.5. Desalination

The use of seawater or other low quality 'brackish' sources for drinking water has a large and specialised literature. At present its use in Europe is very limited, although some regions, such as the Canary Islands, are dependent on it. A model of the water supply in Lanzarote found that the energy intensity was 5.3 kWh/m^3 and the GHG emissions were 3.74 kg CO_2e/m^3 (Meerganz von Medeazza & Moreau, 2007). Similar energy intensity, 4.4 kWh/m^3 was found in La Palma (Sadhwani & Veza, 2008).

Conventional thermal desalination methods are energy intensive and have high GHG emissions: for example, 23 kg CO_2e/m^3 from multiple stage flash (MSF) and 18 kg CO_2e/m^3 from multi effect distillation (MED) (Raluy *et al.*, 2005a). The use of reverse osmosis (RO) can dramatically reduce emissions, for example to 1.8 kg CO_2e/m^3 (Raluy *et al.*, 2005a).

Other studies have estimated the electricity requirements to be 4–8 kWh/m^3 (WssTP, 2011); 6.5–20 kWh/m^3 for thermal desalination, 3.5–7.0 kWh/m^3 for seawater RO and 0.6–1.7 kWh/m^3 for brackish water RO (Vince *et al.*, 2008). In Queensland, Australia, RO desalination was compared with an advanced water treatment plant (AWTP) producing purified recycled water for industrial use from treated effluent with desalination (Poussade *et al.*, 2011). Energy intensity and emissions from desalination were 3.3 kWh/m^3 and 4 kg CO_2e/m^3 compared with and 1.1 kWh/m^3 and 2 kg CO_2e/m^3 from the AWTP. An assessment in the UK found comparable values: 2.2–3.4 kg CO_2e/m^3 depending on the source quality (Reffold *et al.*, 2008).

The use of alternative energy sources could reduce the footprint of desalination. Clearly, the use of standard renewable sources for electricity generation applies equally to other forms of treatment, so does not change the position of desalination relative to those, but solar thermal energy may be a special case. It has been suggested that, where the climate is suitable, it is possible to combine desalination and electricity generation using solar thermal energy (Shinnar & Citro, 2007), though the results are not quantified. A study of California's water supply using LCA (Stokes & Horvath, 2009) found that the emissions intensity of desalinated seawater was 3.9 kg CO_2e/m^3 (using the US national energy mix), 1.5–2.4 times that of imported water (i.e. water conveyed over a long distance). Using solar thermal desalination reduced the emissions to 0.45 kg CO_2e/m^3. When using non-renewable energy, desalination of brackish water produced about 60% of the emissions from using seawater, although the difference was smaller when using solar energy.

A broader but less detailed survey of the opportunities for desalination (Mathioulakis *et al.*, 2007) suggested that solar thermal desalination, or the use of photo-voltaic generation, might be particularly suitable for isolated communities without access to mains water supplies. It also concluded that "conversion of renewable energies, including solar, requires high investment cost and though the intensive R&D effort technology is not yet enough mature to be exploited through large-scale applications."

In summary, current large-scale desalination systems have considerably higher GHG emissions than freshwater treatment systems, but it is the only option in some places and may become viable in others if sources are restricted in future, or as an alternative to long-distance transport (Raluy *et al.*, 2005b). Treating brackish water is generally intermediate between seawater and freshwater. Alternative energy sources, particularly solar, offer the prospect of reducing emissions to a level comparable with freshwater treatment, but are not yet fully mature. However, other environmental emissions, particularly the discharge of concentrated brine, need to be considered in a complete environmental impact assessment.

4.6. Wastewater treatment

It has already been noted that pumping and aeration are energy intensive operations within wastewater treatment. In particular, the activated sludge process has been singled out as a major consumer of energy and source of GHG emissions. Some of the studies give sufficient detail to identify the energy and emission intensities of particular processes. A rough ordering of energy intensity in sewage treatment processes (Hofman *et al.*, 2009) is (low to high): biological (percolating) filters, anaerobic membrane bioreactor, bio-aerated flooded filter, step fed activated sludge, nutrient removal activated sludge, conventional membrane bioreactor. Similarly, for sludge thickening processes: picket fence thickeners, drum thickeners, belt thickeners, belt presses, centrifuges.

In order to produce the Water and Wastewater Energy Best Practice Guidebook for Wisconsin (SAIC, 2006), 95 of the 1010 treatment plants in the state were surveyed. (Note that many of the plants were small: 15% treat 82% of the flow.) For activated sludge plants, which represent the majority, including all the large plants, 54% of the energy was used by aeration, 14% by pumping, 14% by anaerobic digestion and small amounts by other processes. Small plants tended to be less efficient: those treating less than 3758 m^3/day (1 MG/day) using the activated sludge process averaged 1.4 kWh/m^3, and other types up to 50% more, whereas those treating over 19,000 m^3/day averaged 0.6 kWh/m^3, and the upper quartile less than 0.36 kWh/m^3. Estimated savings for best practice included: 10–40% of pumping power, or 50% of secondary treatment, by use of variable speed drives for pumping and blowing; 10–25% of total through operational flexibility; 30–70% of aeration input through process optimisation such as fine-bubble aeration, dissolved oxygen control and variable rate air blowers; 50% of mixing energy through the use of appropriate technology. Further savings were possible through general energy monitoring and management including the use of high-efficiency motors (5–10%).

LCA was applied to a sewage treatment plant in Barcelona with a capacity of 2,000,000 population equivalent, treating 270,000 m^3/day of wastewater (Bravo & Ferrer, 2011). The wastewater line included conventional pre-treatment, primary settler, activated sludge with nitrogen removal, and tertiary treatment. The sludge line (4,000 m^3/day) consisted of thickening, anaerobic digestion, cogeneration, dewatering and thermal drying. Wherever possible, data measured at the plant were used. Direct emissions of methane and nitrous oxide could not be measured and were not included in the assessment. Much of the electricity used was generated on site from a mixture of natural gas and biogas.

The functional unit (FU, for which all the emissions were calculated) was 100 m^3 of thickened sludge (50,000 PE approximately). The total GHG emissions were 5 t CO_2e/FU of which 30% came from the wastewater treatment line and 70% from the sludge treatment line. The largest contributor to the GHG emissions was consumption of natural gas in the cogeneration engine, 2.5 t CO_2e/FU, with mains electricity contributing a further 0.5 t CO_2e/FU for pumps, agitation and centrifuge.

Analysis of a large (25×10^6 m^3/year – www.wastewaterservices.net) wastewater treatment plant at Avedoere, near Copenhagen, found that 44% of energy consumption was used for aerobic treatment (excluding mixing), 18% for inlet pumping and primary sedimentation, 17% for sludge incineration, 9% for mixing in the aeration tanks 6% for pumping stations in the catchment and 5% for outlet pumping (Sharma *et al.*, 2011). It was found that it was possible to reduce the mixing by 50%, saving 0.75 GWh/year, and the plant has been operating continuously at this level since 2007. Based on these values, the overall energy intensity was about 0.67 kWh/m^3, which is comparable to the values found earlier (Table 4) and the reduction was about 0.03 kWh/m^3.

Middleton & Frijns (2010) surveyed several case studies of energy reduction in the wastewater treatment line. The use of advanced process control at Avore in Denmark reduced total energy consumption by 16% from 0.32 to 0.28 kWh/m^3. At Sliedrecht in The Netherlands, plate aerators with high efficiency relative to conventional fine bubble aeration reduced the energy demand for this step by 25%. At Rotterdam, the use of Sharon/Anammox in place of conventional treatment of the liquid from sludge dewatering reduced the energy needed for nitrogen recovery by 50–60%.

An energy benchmarking study on 10 wastewater treatment plants with a variety of secondary treatment systems by the Pacific Gas and Electric Company in California concentrated on secondary wastewater treatment (SBW Consulting Inc., 2001). In general, fixed film processes had lower energy requirements than activated sludge processes. Secondary treatment was responsible for 27–60% of total plant energy consumption (0.13–0.64 kWh/m^3). Ultraviolet disinfection (usually classed as tertiary treatment) required 0.03–0.15 kWh/m^3 for to achieve standard coliform levels, or 0.26 kWh/m^3 where more rigorous discharge permits were in place.

Conventional aerators were replaced by solar-powered, up-flow long-distance circulation units at a 470 m^3/day treatment plant in Pennsylvania (Ambulkar *et al.*, 2011). They resulted in a 47%reduction in mains electricity consumption, confirming the importance of aeration as an energy consumer.

Work in Iran considered energy conservation options, with an emphasis on cost saving, at several stages in both the wastewater and sludge lines (Ataei, 2010). The reference plant treated 10,000 m^3/day of wastewater, using a conventional activated sludge process. The baseline energy intensity appears to have been about 0.5 kWh/m^3, or a total of 5 MWh/day. Pumping and aeration have previously been identified as two major consumers of electricity. Two options for improving these were considered: variable-frequency drives (VFDs) to adjust the motor speed to the demand and more efficient motors. Using VFDs reduced energy use by 45% in appropriate applications, with a short payback period and reduced maintenance costs, but they are not suited to applications where the flow is constant. Enhanced efficiency motors below 100 hp were typically 5% more efficient than standard ones, with payback periods less than one year, but the difference in efficiency for larger motors was less than 2%, with proportionally longer pay-back periods. Other options considered were maximising the use of biogas, which is covered by several other studies, the use of micro-bubble aerators and cooling tower designs for the discharged liquid. The results from these are difficult to put into the context of the complete system.

Life cycle analyses were carried out for two plants serving Oslo over the period 2000–2007 (Venkatesh & Brattebø, 2011). One (VEAS) used aerobic biofilters for primary wastewater treatment, while the other (BEVAS) used both aerobic and anaerobic biofilters. Both used anaerobic digestion of the sludge, but only BEVAS used the biogas for electricity generation. The population served was 50,000 in 2002 and 60,000 in 2007, but the volume of wastewater treated changed relatively little: from 119×10^6 m^3 to 112×10^6 m^3, a fall of some 15% in the per capita volume treated. From 2001 onwards the electricity consumption was fairly constant at 28 to 30 GWh/year, which came from hydropower. On-site generation from biogas rose from 7.1 GW hour to 9.52 GW hour. Due to the use of hydropower the energy related GHG emissions were slightly negative (about -5 g CO_2e/m^3) so emissions related to chemical use had the biggest impact (1–1.9 kg CO_2e/m^3). When it was analysed using the Nordic electricity generation mix (22% fossil fuel), the emissions related to chemicals were little changed due to the substitution of urea and superphosphate with sludge fertiliser and ammonium nitrate, but energy-related emissions rose substantially.

Some novel, low energy, alternatives to standard aerobic treatment were advocated by Shilton *et al.*, 2008, though not fully quantified. Waste stabilisation ponds (WSP) are uncovered, aerobic algal ponds, which have been used for community-scale wastewater treatment. The construction costs are relatively low and there is no direct energy input, or a low input for mixing in high-rate ponds. They suggest that sunlight can provide disinfection in WSP, although this may depend on consistently high levels of UV-B light, which would not be found at higher latitudes. Small-scale anaerobic ponds have been used in France and Australia to produce biogas, less efficiently than conventional digesters, but at lower cost. Algae can be harvested from high-rate ponds for as biomass for digestion (Heubeck *et al.*, 2011). The yields may also be high enough in warm temperate climates for this to be possible with WSPs. At present these methods are not fully developed for large-scale treatment works, but are proposed as low-energy alternatives for a variety of applications.

Hofman *et al.* (2009) surveyed previous studies of methods to recover energy from wastewater. They concluded that the best option was drying undigested sludge after pre-settling using waste heat, followed by incineration/co-firing (presumably with heat recovery). The available conference presentation does not quantify the results.

4.7. Sludge treatment

The sludge treatment line was examined in detail by Barber (2009), who used a model to assess the carbon footprints of different sludge treatment and disposal options. Raw, digested and advanced digested sludges were considered, based on data collected from 24 plants in the UK. The author had previously shown that the GHG emissions saved by utilising biogas from primary digestion were offset by the methane emissions from open secondary digestion tanks in typical UK designs. These could be reduced by collecting a flaring the methane, thereby converting it to short-cycle carbon-dioxide. Pretreatment, such as hydrolysis could reduce the carbon footprint to, or below, zero. The options included in the model included: no digestion, primary and secondary (open and closed) digestion, and pre-treatment with primary digestion. In each case, the final stage was dewatering, and possibly drying, before incineration, co-firing or land application (with lime). The carbon footprints were calculated to the plant gate and including the disposal option.

The carbon footprint of dewatering was 0.04–0.05 t CO_2e/t DS for digested sludge and 0.1 t CO_2e/t DS raw sludge, of which 59% was power use for liquor treatment, 32% power use for dewatering and the remainder associated with chemical production. If the sludge was applied to land, transport and spreading, including nitrous oxide emissions, had a footprint seven times that of dewatering. It does not appear to include the offset which accrues from the displacement associated with chemical fertiliser use. The use of electricity and gas for sludge drying also had seven times the carbon footprint of dewatering.

Without drying, the plant operating emissions were about 20 kt CO_2e/year without digestion, 15 kt CO_2e/year with open secondary digestion, -1 kt CO_2e/year if the methane from secondary digestion was flared and -5 kt CO_2e/year using advanced digestion (not clearly defined). Drying had an emission intensity ranging from 0.05–0.15 t CO_2e/t DS for advanced digestion to about 0.6 t CO_2e/t DS for raw sludge or standard digestion. This increased the plant emissions to 5–35 kt CO_2e/year depending on the treatment option. The overall carbon footprint for treatment and land spreading ranged from 0 with advanced digestion to 40 kt CO_2e/year for raw sludge. If drying was included, the footprint for each option was increased by 10–20 kt CO_2e/year. The footprints for drying and incineration were similar. The lowest footprints – negative in most cases – were obtained for incinerating dried sludge pellets for energy reuse.

An LCA study in Switzerland also focused on final sludge treatment and disposal options, for a 300,000 person equivalent (90,000 m^3/day) plant (Houillon & Jolliet, 2005). Of the six options considered, incineration (co-fuelling) dried sludge in cement kilns had the lowest GWP (75 kg CO_2e/t DM), despite the use of gas for drying, due to the fossil fuel use displaced. Landfill disposal resulted in 1300 kg CO_2e/t DM, of which half was due to methane emissions. Agricultural land spreading of limed sludge was the next largest emitter: about 500 kg CO_2e/t DM, of which 313 kg CO_2e/t DM is the net result of lime production (less lime application displaced) and including a significant contribution from methane emissions. The assessment also accounted for substitution of artificial fertilisers (ammonium nitrate) by sludge. Pyrolysis (300 kg CO_2e/t DM), fluidised bed incineration (150 kg CO_2e/t DM) and wet oxidation (150 kg CO_2e/t DM) had lower net emissions: process inputs of gas and electricity were partially offset by gas production or waste heat recovery.

A similar study considered three options for disposal of sludge from a treatment plant in Spain (Hospido *et al.*, 2005). Currently the sludge is digested anaerobically (30% of biogas used for heating only; remainder flared), dewatered and applied to agricultural land, but resistance to this is increasing, particularly due to limits on heavy metals and other chemical loads. This option had net GHG emissions of 200 kg CO_2e/FU (FU = 1 t thickened mixed sludge, dry basis), of which 70% were emissions from land spreading. Incineration of undigested sludge in a fluidised bed after mechanical dewatering increased the net emissions to 250 kg CO_2e/FU, of which 75% were from the incinerator. Pyrolysis following mechanical dewatering and thermal drying had net emissions of 620 or 450 kg CO_2e/FU depending whether syngas only or all the products were used for energy recovery. These results revers the order found by Houillon & Jolliet. The difference may be partially explained by the other changes to the sludge processing, the omission of liming in land application and the lack of energy recovery from incineration.

A different range of final treatment and disposal options for digested sludge were considered for a plant in Stockholm (Johansson *et al.*, 2008). Three options led ultimately to land application: direct use for restoration after mining, composting with other materials for use on golf courses and storage followed by application to agricultural land. All of these had similar net GWP about 70 kg/person^{-1}year^{-1}, dominated by the GHG emissions after application. Supercritical water oxidation with phosphorous recovery had net emissions of -1.7 kg/person^{-1}year^{-1}, due to the elimination of direct GHG emissions and the recovery of heat from the process.

4.8. Recycling and reuse – closing the loop

At present, the water-wastewater system operates as an open loop with the environment as the source and sink. Water supplies are extracted from surface water and groundwater, often at considerable distances from the point of use. Most municipal wastewater is treated at points relatively close to urban centres and discharged into surface water, including oceans. In some regions, particularly where supplies are scarce, this loop is being (partly) closed by using treated wastewater, though not generally as a source for potable water. This potentially saves energy for extraction and conveyance. For example, in southern California the energy and emissions intensities for recycled water were comparable to water imported from the north of the state because of the high energy requirement for conveyance (Stokes & Horvath, 2009).

One of the key concerns is hygiene, because municipal wastewaters contain micro-organisms, such as faecal coliforms, that are adapted to the human gut. Other contaminants, such as artificial chemicals, are present in much higher concentrations than normal freshwater sources. The use of microfiltration in membrane bioreactors may provide a cost-effective method of treating to non-potable quality; including ultrafiltration can produce potable water (Bennett, 2006). However, there may still be public resistance to the concept of using recycled water (Jeffrey, 2002).

Using recycled water for industrial supplies could relieve the pressure on freshwater sources for potable water. For example, a wastewater treatment plant in the east of England (the driest area of the country) supplies 1200 m^3/day to a nearby gas-fired power station. The effluent undergoes three stages of treatment: screening to remove particles over 150 μm, microfiltration to remove particles and microorganisms down to 0.1 μm and reverse osmosis, which removes 80-90% of dissolved ions. In addition to saving 1200 m^3/day previously drawn from the potable supply, it has reduced the load on the power station demineralization plant (Murrer, 2002).

A larger plant in Melbourne is intended to supply 2.5×10^6 m^3/year (6850 m^3/day) for use by a plastics manufacturer and irrigation of amenity land (Anon, 2011). It will use ultrafiltration followed by single-pass reverse osmosis for irrigation or two-pass reverse osmosis for manufacturing use. In California, a recycled water supply started in 1969 now provides 50,000 m^3/day for landscape irrigation, agriculture, toilet flushing in office buildings, cooling towers and industrial processes through one of the largest dual supply networks in America (ACWA, 2011).

Some of the results reviewed in previous sections included the energy and GHG implications of recycling. A general estimate for Europe gave 1–6 kWh/m^3 for recycled water (WssTP, 2011). In California, Stokes & Horvath (2009) estimated 1.1 kg CO_2e/m^3 for imported water and 1.0 kg CO_2e/m^3 for recycled.

The use of recycled water in drought-prone areas of Spain was considered in an LCA study (Pasqualino *et al.*, 2010). They noted that the tertiary treatment for non-potable use (flocculation, pre-chlorination, sand filtration, UV treatment and post-chlorination) added only 0.16 kg CO_2e/m^3 to the emissions of 0.83 kg CO_2e/m^3 from wastewater treatment (primary, secondary and sludge). The net GHG emissions if the treated water was used to displace potable water were 0.71 kg CO_2e/m^3. If it replaced desalinated water the net emissions were −2.12 kg CO_2e/m^3.

The most systematic planned use of recycling is part of the design for ecocities in Qingdao (China), operating in clusters of 1500–2000 people (Novotny, 2011). The design assumes individual consumption of 130 l/day, of which only 50 l/day is supplied from the grid. Grey water is treated by sand filtration, membrane filtration, reverse osmosis and ultraviolet disinfection before being returned for non-potable reuse. More contaminated wastewater ('black' water) passes through settling, anaerobic treatment, membrane filtration and ultraviolet disinfection. It is then held in surface storage before being fed through the grey water treatment process. The total emission intensity is estimated to be 1.6 kg CO_2e/m^3, which is comparable to the sum of water and wastewater treatment emissions found in Europe. Systems of this complexity would only be possible as part of substantial new developments, so are unlikely to make a big impact on demand in heavily urbanised countries with stable populations.

The majority of authors do not see recycled water as a direct source of potable water, which would require treatment to higher levels than most of them have considered. However, by displacing the use of potable water in industry or amenity irrigation effectively increases the availability of freshwater for potable use. The net effects on the carbon footprint are generally small, except where energy-intensive sources, such as long-distance conveyance or desalination, can be displaced.

In addition to treated wastewater, other sources may be available for low-quality use. These include rainwater, grey water, surface water and sustainable urban drainage systems (SUDS) (Caffoor, 2010). There is some potential to provide limited treatment at a local level, to avoid the use of energy to transport them to and from treatment works. Household rainwater harvesting and grey water recycling systems were considered as demand management options in Section 0, but current evidence is that they are likely to increase carbon footprints. Community-scale systems may offer economies of scale to offset this. Urban surface waters and SUDS may be contaminated with a range of chemicals, including vehicle emissions and pesticides used on pathways or amenity land. The possible applications without full treatment are therefore limited. Local reuse may provide another method to displace the use of potable water, but it will not necessarily reduce the carbon footprint with current technology.

5. POTENTIAL FUTURE CONFIGURATIONS FOR LOW EMISSIONS

Several authors have set out visions for low energy, low emission, or even zero net emission future configurations for the water industry (e.g. Caffoor, 2010; Crawford, 2010; Frijns, 2012; Hofman *et al.*, 2009, 2011; Novotny, 2011; WSAA, 2009; van der Hoek, 2011). The following sections draw heavily on these and on supporting data from the other studies reviewed above. A common theme to several of the visions is the need for an incremental approach: a series of 'wedges', each delivering a component of the saving. Some of these can be achieved relatively quickly, with short pay-back periods for investment ('low-hanging fruit'), whereas others imply substantial changes to capital assets over much longer periods. To meet its commitments, the industry needs to find sufficient of the first type to achieve a 20% (26% in the UK) emission reduction on a 1990 baseline by 2020.

As some of the results from Norway (Venkatesh & Brattebø, 2011) and Canada (Bonton *et al.*, 2012; Racoviceanu *et al.*, 2007) have shown, the assessments can change dramatically when electricity sources with low GHG emissions, such as hydro or nuclear, are used. The planned reduction in dependence on fossil fuels, or the use of carbon capture and storage, in other countries should help to reduce the carbon footprint of the water industry, but this discussion will focus on changes within the industry itself.

5.1. Reducing emissions from water supply

Demand management was discussed in Section 0. In principle, reducing demand has a proportional effect on operational energy use and emissions throughout the supply network. However, it has to be set against a background of increasing demand (Caffoor, 2010; Warren & Holman, 2012). In the UK, it is being promoted by the Government (Defra, 2008), water companies and NGOs, but the outcome is highly uncertain. The targets for domestic use in the UK require a 20% reduction in consumption, but provide no specific incentives to consumers to achieve it. Although water efficiency is part of new building codes (DCLG, 2006), the turnover of housing stock is very slow (about 0.5%/year) so the effect will only be measurable over decades. The turnover of appliances in existing houses is quicker, but the unit benefits are smaller, so the benefits will also be slow to emerge. It has already been noted that grey water systems at household scale have limited scope outside new construction and entail their own GHG emissions, so it seems unlikely that they will deliver substantial reductions in the short term. It has been estimated that the introduction of universal metering would reduce domestic demand by 10% by encouraging behavioural change and the adoption of efficient appliances, but this is not due to be complete until 2030. Some companies have low usage tariffs, but the thresholds are too low to provide an incentive to most households (e.g. 205 l/day for one company). It therefore seems likely that any reduction in consumption by 2020 will be less than 10%, but 20% should be achieved before 2050. Any reduction would produce a proportional reduction in the operational emissions throughout the supply chain, but have a much smaller impact on wastewater treatment.

Achieving further reductions in those countries that are seen as a model for the UK will be more difficult, so any changes are likely to be smaller. The greatest potential for savings is in the countries with very high demands, but there is little literature available in English to indicate whether they are pursuing demand management strategies and whether they are likely to succeed. Some of the recent accessions to the EU currently have relatively low consumption. There is a risk that the demand for water will increase as their economies converge with the existing members.

There are significant opportunities to improve the efficiency of conveyance and distribution, estimated to be 30–50% of energy consumption (Middleton & Frijns, 2010). Some of the benefits arise from optimisation of operations, whereas others require asset replacement, which will normally only occur as part of the normal cycle. Therefore, the full benefit will only be achieved over the medium term. Within such as strategy, long distance and high-lift transport should clearly be avoided.

Pumping within water treatment works was shown to be a major component of energy demand, sometimes the majority of consumption, in several studies, but the proportion varied widely according to the treatment system in use. The sources quoted above indicate that reductions of 5–10% are possible by using more efficient pumps and that up to 45% reductions may be made through the use of variable speed drives where the required flow is variable. Some of these improvement could also be applied to conveyance and distribution.

Many of the studies identified GAC filtration as one of the largest source of emissions when renewal of the GAC was included. The most direct comparison of an alternative was given by Bonton *et al*. (2012). In that case a hypothetical conventional-GAC plant was compared with a nanofiltration plant producing drinking water. The GHG emissions from the nanofiltration plant were 40% lower than the conventional plant when using US generation mix and even more favourable when using hydro-power.

This was the only case study of any kind to attempt a like-for-like replacement, so its wider applicability is uncertain. Indeed some plants include both GAC and nanofiltration to remove different contaminants. However, it does suggest that lower emission alternatives to GAC filtration may be suitable for some situations.

None of the literature offered a single technology that was likely to produce a dramatic reduction in the total energy requirements for treatment in all situations. One reason is the need to use several processes to remove large and small particles, bacteria, viruses and both organic and inorganic chemicals. The choice of processes will depend on the source water being employed, so water treatment works will typically have options available for all the contaminants they encounter, even if these occur sporadically, to ensure that they remain in compliance with the Drinking Water Directive.

The chemicals used in water treatment have a substantial carbon footprint. It is important to include this component, since reducing the direct use of energy might increase chemical consumption and hence the carbon footprint. (This is the reverse of the historical effect of substituting ozonation and ultra violet sterilization for chemical disinfection.) Thus the ideal processes would use fewer chemicals and less energy to achieve the present results.

The LCAs that included construction and decommissioning generally found that these were small (5–15%) or negligible due to the long lifetime of capital assets in the water industry. This simplifies the assessment of the carbon footprint, but emphasises the fact that major changes will take decades to complete.

In principle, the WFD should lead to an improvement in the quality of surface water sources. In theory, this could reduce the need for treatments such as GAC filtration if the load of dissolved organic chemicals fell substantially. The ability of existing measures to deliver an improvement is unproven, so it is not yet possible for water companies to plan on this basis, as they must supply water to rigidly defined standards. In the case of groundwater, residence times are so long that it may be several decades before any changes become clear. Given the uncertainty and the time scale, it is not prudent to assume that there will be any reduction in GHG emissions as a result.

Conversely, in many parts of Europe, climate change will probably reduce the volume of surface water available. This is likely to lead to an increase in the use of lower quality sources. These may be simply be existing sources with lower flows, but the same chemical and microbial loads, or alternative sources: brackish water, seawater and recycled water.

These were discussed in Sections 4.5 and 4.8. The GHG emissions from desalination were always higher than from using freshwater sources, unless long-distance conveyance was avoided. The main benefit of recycling was to replace potable water for industrial and other uses, but not reduce emissions. Modern desalination technology has considerably reduced the energy required for desalination and it is likely that this trend will continue, but it will remain more energy intensive than using freshwater, because any dramatic improvement in technology could probably also be applied to freshwater treatment. Seawater desalination is only applicable to communities in coastal areas, otherwise the energy cost of conveyance has to be added, so its impact on a European scale will be limited.

The direct use of solar thermal energy for desalination is most applicable to small-scale treatment. Indirect use of solar energy via electricity generation is not specific to the water industry and forms part of the general shift to renewable sources.

Making some very broad assumptions based on this discussion can provide a crude estimate of the plausible total energy reduction. Reducing demand by 10% would reduce total operational consumption by the same amount. The lower end of the distribution efficiency improvements suggested by Yates & Weybourne (2001) was 30%. Pumping was the major consumer of energy in most treatment works, unless ozonation and GAC production were included. Various improvements to pumping have been suggested, including 5–10% by more efficient pumps and up to 45% in some applications for variable speed drives. Other improvements to treatment processes have been less well quantified, but some delivered up to 40% reductions for the complete process, including pumping. Let us assume that an average of 20% is achievable for the plant as a whole, by a combination of measures.

5.2. Reducing emissions from wastewater treatment

There are four main components to be considered within wastewater treatment: general efficiency improvements, including pumping; aerobic treatment, including activated sludge; anaerobic digestion; and sludge disposal. The last two will be discussed in the subsequent sections.

Several of the studies reviewed in Section 4.6 showed that pumps and other motors were significant components of energy use and GHG emissions sewerage and wastewater treatment, though not as large as in the water supply. Estimates included 14% of energy for on-site pumping only (SAIC, 2006), 10% of emissions for pumps, agitator and centrifuge (Bravo & Ferrer, 2011) and 39% of energy for catchment pumping, discharge pumping and on-site pumping and mixing (Sharma *et al.*, 2011). Efficiency gains of 10–40% were possible (SAIC, 2006). In addition, general operation improvements could save up to 25% of total energy inputs (SAIC, 2006).

In almost all cases, the major consumers of energy, and therefore the main targets for improvement, were aerobic treatments, especially the activated sludge process. Typical values were 54% (SAIC, 2006) and 44% (Bravo & Ferrer, 2011). Some of these showed that it was possible to improve the efficiency of aeration considerably. For the large activated sludge plants surveyed by SAIC (2006), the average energy intensity was 0.6 kWh/m^3 and that for the upper quartile of efficiency was 0.36 kWh/m^3, offering a potential 40% improvement if no other factors were involved. They also found that the aeration input could be reduced by 30–70% through improvements, such as fine-bubble aeration, dissolved oxygen control and variable rate blowers. Specific improvements included 16% of total energy by advanced process control, 25% of aeration energy by using high efficiency plate aerators (Middleton & Frijns, 2010) and 50% of mixing energy in aeration (which originally constituted 9% of the plant total) (Sharma *et al.*, 2011). Another estimate is that optimisation and better process control could reduce the energy used in the process by 50% and that further process monitoring and control throughout the plant could save a further 10–20% of total energy (Georges *et al.*, 2009). Wastewater treatment plants should adopt approaches to process control that are already common in the chemical engineering and manufacturing industries, such as statistical process control, implementation of continuous improvement ('Kaizen') and application of 'six sigma' (Stephenson, T. in Ainger *et al.*, 2009).

As most of these proposals involve modification or better control of existing processes, they would be feasible on relatively short time scales and comparatively low cost. The magnitude of the reductions suggested would meet most or all of the reduction in emissions required by 2020.

However, a more radical proposal, with greater potential benefits, is to replace aerobic wastewater treatment with low-temperature anaerobic processes.

> "[By 2050] there will have been a revolution in the core unit operations used in municipal sewage treatment works. A major development will have been the application of anaerobic processes to mainstream flows. Ambient temperature anaerobic treatment of sewage will be possible by fortification of the influent waste stream, either from sludges generated on-site or other imported organic wastes. These processes produce biogas to recover energy, but the major benefit will be reduced aeration costs." (Stephenson, T. in Ainger *et al.*, 2009)

> "The major change to wastewater processing will be the move to low temperature anaerobic treatment and the use of anaerobic membrane bioreactors…" (Caffoor, 2010)

Some of these processes have been in use for some time, such as the upflow anaerobic sludge bed (UASB), which is suited to wastewaters with lower total solids contents (3% TS) than the digesters conventionally used for sewage sludge (Lettinga, 1995). At ambient temperatures above 15°C, no additional heating is required; below this, some of the biogas produced will be used for heating. Unlike the activated sludge process, only the treated liquid effluent stream needs to be aerated, which can halve the energy required. Alternatively, further anaerobic processes designed for low-strength wastes can be employed.

It has been estimated that by 2030 a similar flow sheet could replace aerobic treatment consuming 0.15–0.7 kWh/m^3 with anaerobic treatment producing 1.7 kWh/m^3 (WssTP; 2011GWRC, 2010). The Water Environment Research Foundation in Canada also proposed treatment plants based largely on anaerobic processes, with membrane-based treatment for the liquid effluent (Crawford, 2010).

Although a move to anaerobic treatment is the dominant proposal for low-emission wastewater treatment, an alternative approach was suggested by Hofman *et al.* (2009). This consisted of an optimised primary treatment step, including phosphorous recovery, followed by a dynamic membrane reactor (Liu *et al.*, 2009), then nitration and sand filtration for the liquid effluent. The sludge would be dried using waste heat and used as a fuel source for incineration or co-firing (for example) a cement furnace. This design has not yet been tested in practic

5.3. Energy from water and wastewater treatment

Although the process improvements suggested in the previous sections could substantially reduce the energy consumption of water and wastewater treatment, better energy recovery has the potential to reduce it to zero, or even to become a net producer of sustainable energy (e.g. Heubeck *et al.*, 2011; Hofman *et al.*, 2009; McCarty *et al.*, 2011; Novotny, 2011).

The opportunities in water treatment are comparatively limited and many have not been properly quantified. It has been noted that a small fraction of the emissions from groundwater treatment are methane released by degassing (Frijns, 2012). Recovery of these emissions is under development at a drinking water plant in the Netherlands (van der Hoek, 2011). Some low-grade heat recovery may also be possible after treatment (van der Hoek, 2011).

Larger-scale electricity generation is possible by the use of turbines within gravity-fed flows, for example from reservoirs prior to treatment. For example, the UK currently has an installed capacity of 9 MW (Howe, 2009). The capacity could be greatly increased by the development of low-head, low-flow turbines, such as Archimedes screws. A 180 kW example is being commissioned in the UK by Yorkshire Water (Howe, 2009). Clearly this approach is only appropriate for some situations: ground water extraction will remain an energy consumer.

Micro-turbines to generate electricity from flows in pipes, even within the treatment works are also proposed (SAIC, 2006; Howe, 2009), but these are likely to have limited applications (Georges *et al.*, 2009).

As landowners, water companies also have the opportunity to exploit wind generation, particularly around rural reservoirs (Howe, 2009). Public opposition can be a problem where there is nearby housing or the site has a high amenity value for leisure activities or as a beauty spot. Provided permission can be obtained, this could help to offset the demand (when operating) or feed in to the grid.

Many wastewater treatment works already include capture biogas from anaerobic digestion of sludge, but many only use it for heating and flare the rest (e.g. Hospido *et al.*, 2005). Although this reduces the GWP by converting methane to short-cycle carbon-dioxide, it wastes a valuable source of renewable energy.

Combined heat and power (CHP) units use biogas-powered engines to drive generators and recover the waste heat for process heating. Currently, for example, South-West Water in the UK send one-third of their treated sewage sludge to anaerobic digestion, from which they generate 10.6 GWh/year, supplying 4% of their total power (Crawford-Brown, D in Ainger *et al.*, 2009). They thus have the potential to generate 12% of their needs with the treatment existing processes. In the Netherlands, 33% of the power used in wastewater treatment is already generated from biogas (Hofman *et al.*, 2011). A CHP unit installed on a treatment works in Romania was able to generate 58% of the plant's power requirement (ManagEnergy, 2004). Similarly, an installation in Portugal reduced the electricity and natural gas consumption by 67% and GHG emissions by 39% (ManagEnergy, 2008). At a UK treatment works, advanced digestion reduced the plant's footprint from 20 kt CO_2e/year to -5 kt CO_2e/year. In 2008 Waternet (Amsterdam) claimed a GHG emission reduction of 13 kt CO_2e by sustainable power generation, and 3 kt CO_2e by sustainable heat generation from biogas. Heat from the combined incineration of digested sludge and municipal waste provided sustainable heating to 4,000 houses by a district heating network (Mol *et al.*, 2011). Two wastewater treatments works in Austria using the activated sludge process and anaerobic digestion are reported as being self-sufficient in energy (Nowak *et al.*, 2011). The efficiency of biogas production can be increased by using mixed waste, for example including waste from the food sector, to optimise its carbon:nitrogen ratio.

These examples illustrate that energy production to offset a proportion of consumption is readily achievable and that very substantial reductions, even down to zero, are possible with current technology. If plants convert from aerobic to anaerobic treatment, the quantity of biogas available will increase while their demand for energy decreases, allowing them to become net exporters of energy. In future, hydrogen fuel cells could provide a more efficient alternative to traditional generators (Howe, 2009). Microbial fuel cells, currently at the experimental stage, may allow simultaneous treatment and power generation (Puig *et al.*, 2011).

If the wastewater sector of the water industry becomes a net producer of energy, it will be able to offset more of the remaining consumption by the supply sector. This should help to reduce the combined emissions to the desired level.

5.4. Sludge disposal and energy

As noted in Section 2.1, there are four common disposal routes for sewage sludge: landfill, agricultural use (including composting), incineration and as a fuel. Landfill is still widely used in some countries (Figure 17), but it has causes very high GHG emissions (Houillon & Jolliet, 2005) and should be phased out under the EU Landfill Directive (1999/31/EC). Agricultural use also produced substantial GHG emissions (Houillon & Jolliet, 2005;Barber, 2009).

Several of the studies reviewed in Section 4.6 included incineration of the sludge or using it for co-firing after dewatering, and possibly drying (Barber, 2009; Hofman *et al.*, 2009). In some cases, for example the fluidised bed incinerator reported by Hospido *et al.* (2005), incineration produced GHG emissions, because it was purely a means of disposal, not a source of energy. In the detailed investigation by Barber (2009), incineration alone slightly increased the total emissions compared with the plant emissions, although it was preferable to land spreading. Co-firing the sludge resulted in negative total operating emissions for most of the treatment options considered, particularly advanced digestion. Similarly, Houillon & Jolliet (2005) found that using the sludge to co-fire a cement kiln resulted in a negative total footprint. An assessment of five co-combustion scenarios found that all produced net energy gains of 0.58–5.0 kWh/kg dry solids and most produced a reduction in GHG emissions (Cartmell *et al.*, 2005).

Note that some of these assumed the use of undigested slurry as the fuel. Minimizing the direct use of energy by current treatment works would require digestion for biogas production. The options for the use of sludge from fully anaerobic processes have not been evaluated.

5.5. Adaptation option maturity

Many of the technologies and interventions discussed in the preceding sections are already being deployed across the European water sector. Others remain simply near-market potentials and yet others are at the pilot or trial stages of development. Whilst not comprehensive in its coverage of options, the content of this report does provide an opportunity to compare and contrast the major energy and GHG emission reduction strategies in terms of their relative maturity (Table 6: Adaptation option maturity

Table 6: *Adaptation option maturity*

ADAPTATION OPTION	MATURITY	COMMENTS
Increased use of hydropower (inc. micro-turbines)	High	Geographical limitations on application
Demand Management	Med - High	Techniques further developed in some states than in others
Operations optimisation	High	
Asset replacement	Medium	Dependent on asset replacement cycles - medium term gain only
More efficient pumping (e.g. variable speed drives)	High	
Replacement of GAC processes	High	
Sustainable energy driven desalination	Low	
Reduced aeration in WW treatment	Med-High	Variable potential for savings
Use of low temperature anaerobic processes (inc. biogas capture and use)	Medium	
Incineration of sludge	Medium	Overall energy and GHG value debatable

5.6. Visions of the future of urban water

Several authors have advocated the need to consider the sustainability of water and wastewater for several years, including global warming and all other aspects of its environmental impact (e.g. Hellström *et al.*, 2000). Differing visions of the future have been produced, but most focus on issues of demand management and local environmental impacts without a thorough analysis of greenhouse gas emissions from the full life cycle including construction, operations and decommissioning.

Among the papers reviewed above, those based on life cycle assessment gave the fullest consideration to the total environmental impacts, although we have concentrated on greenhouse gas emissions. Some of them show that different environmental concerns may come into conflict. For example, the case of desalination or long-distance conveyance in

Spain (Raluy *et al.*, 2005b), where the former had lower GHG emissions, but caused the discharge of concentrated brine. However, these papers were generally comparisons of specific options for treatment, not complete visions for the industry as a whole.

The dominant theme of these visions for water is decentralisation, involving local supplies, where possible, local wastewater treatment and maximising recycling. This was identified as a key driver by the WssTP:

> "Identify the longer term paradigm shift, which could involve greatly different decentralised infrastructure models, and the provision of water of varying quality but fit for purpose." (WssTP, 2011)

The final report of the EU Sustainable Water Management in the City of the Future (SWITCH) project says

> "By adopting an approach to wastewater management that is based on decentralised solutions for separation and reuse, the key health and pollution control objectives are achieved as well as the following additional benefits: increased access to sanitation, water savings, flexibility to change, recycling of nutrients, financial savings, employment generation, energy recovery, more efficient treatment and an increase in urban biodiversity and amenity." (Howe et al., 2011)

Similarly, Caffoor (2010) identified local supply treatment and wastewater processing with recycling, utilising micro-generation as the typical model, with bulk supply of potable water as the exception:

> "A typical community level water supply process flow-sheet will probably consist of: low quality (non-potable water) supplied, either via large raw water mains or sourced locally, to the household/community; and treatment provided by end of tap or community level packaged treatment. In a few scenarios bulk supply of potable water may be identified as the most sustainable option. This will be consequent on DEFRA's reviewing water quality standards. Local sources of water will be used for low quality applications, as will rainwater, grey-water, surface water, and water contained in Sustainable Urban Drainage Systems (SUDS'). Local processing will be powered by micro-generation of electricity." (Caffoor, 2010)

The Water Services Association of Australia raised a note of caution about planning that does not integrate water and wastewater

> "One simple example illustrates this point. Water conservation measures, particularly those targeting indoor water use (such as water efficient washing machines and shower heads), not only reduce the volume of water consumed but also reduce significantly the volume of water discharged to the sewerage system. Recycled water projects rely on reliable flows in the

sewerage system. In some instances around Australia, the volumes of sewage have dropped by up to 40 percent, which reduces the yields available from recycled water schemes." (WSAA, 2009)

New, planned developments as envisaged above would embody this integration, but attempting to retrofit piecemeal solutions into existing communities could easily produce unintended consequences. Even without community-scale recycling, substantial and successful water conservation and grey water re-use could have a significant effect on the physical, chemical and microbiological characteristics of wastewater, with implications for sewerage and treatment.

All of these reports lacked a quantified assessment of the impact on energy and greenhouse gas emissions. It is unsafe to assume that localisation and micro generation will produce reductions without a full assessment. Indeed, it has already been noted that grey water treatment at the scale of single buildings, including houses, office buildings, medium-sized hotels and small secondary schools is likely to increase emissions (Parkes *et al.*, 2010). Some processes have intrinsic efficiencies of scale. For example, the proportion of the energy produced by biogas plants that has to be used to heat the sludge decreases with volume.

An assessment in Sweden compared a conventional, centralised GAC plant supplying drinking water to flats in Göteborg with local membrane systems using one-step ultrafiltration or two-step microfiltration plus reverse osmosis for the potable water only (Westrell *et al.*, 2002). In each case, pumping accounted for a large proportion of the energy use. The local one-step membrane system had the lowest process energy requirement; the other two systems were similar. However, the microbial risk assessment found that the risks of exposure to certain pathogens were above the accepted thresholds when using the conventional and one-step membrane systems. The two-step membrane reduced the risk considerably.

Another project in Sweden considered alternatives for wastewater treatment. A water supply and wastewater supply for 15,000 inhabitants in a new development in Stockholm was assessed prior to implementation (Jeppsson & Hellström, 2002). A centralised treatment plant designed for good environmental performance was compared with a system using more local treatment and separation of wastewater streams, but not recycling. Both systems showed lower environmental impacts than standard systems at the time and were net energy producers, though the source-separated system produced 10% less net electricity and 23% less net heat. Each performed better than the other on some of the environmental criteria. For example, the source-separated system discharged 40% less nitrogen to water, but 34% more phosphorous.

Unfortunately, neither of these dealt with an integrated water-wastewater system or evaluated the full implications for greenhouse gas emissions.

It is important that planning integrates the security of supplies with a full assessment of the energy, greenhouse gas and environmental impacts. A few studies have begun to do this Novotny (2011) outlined an energy CO_2 (not GWP/CO_2e) balance for an ecoblock. It was an illustrative example, rather than a complete assessment, which showed the total CO_2 emissions from a population of 100,000 fell from 70 t/day to 30 t/day as per capita delivered from the grid decreased from 500 l/day to 200 l/day, mainly by water conservation, then rose to 50 t/day at 40 l/day delivered as the level of recycling increased. This design therefore failed to achieve carbon neutrality, but it illustrated what is possible with current technology.

An assessment of water services for a green field development for 86,000 inhabitants near Melbourne included a life cycle assessment of four options: (A) conventional, (B) reclaimed water, (C) recycled storm water and (D) grey water with treated rainwater for potable use (Sharma *et al.*, 2009). The ranking by greenhouse gas emissions, from lowest (best) to highest, was D, B, A, C. Option D also had the lowest eutrophication potential and solid waste generation, but the highest life cycle cost and the lowest reliability. A related paper noted that

> "...the current level of urban water service provision can't be provided within the existing centralised system framework, unless there is a significant increase in investment, to enlarge and rehabilitate the existing centralized systems... Decentralised and distributed water and wastewater systems, which are planned within an integrated water management concept, are being promoted either in combination with centralised systems; or alone as the sustainable solution for urban water servicing. Current urban water systems are beginning to undergo a transition, where decentralised systems will play a major role in the long-term sustainability of these systems to meet the above mentioned challenges." (Sharma et al., 2010)

It cautioned that further investigation is needed:

> "However, since decentralized systems are relatively new and involve increased complexity there are wide knowledge gaps in their planning, design, implementation, operation and management, which are impeding their uptake."

These studies show that decentralised systems may be able to compete with conventional systems on most environmental performance measures, although the comparisons have not been made with centralised systems that have been optimised for energy production and greenhouse gas minimisation, as discussed in previous sections. Conversely, local treatment and recycling are relatively novel and their performance can be expected to improve in future. The methods of assessment that are being developed in these studies, using LCA to estimate the complete greenhouse gas emissions and other environmental impacts together with assessment of the financial and social costs should form the basis for informed discussion of the future provision of water and wastewater services.

5.7. Governance in pursuit of a low carbon water sector

Today, there is no doubt that the major challenge facing urban water services is to achieve the Millennium Development Goals set at the 2002 Johannesburg summit, one of which is to provide access by 2015 to water and sanitation for billions of people in developing countries who at that time did not have it. But for the present we are not concerned with such a formidable problem whose solution, in the opinion of the experts (e.g. Falkenmark, 1998), requires a Herculean effort, but rather with those general ones arising from the rapid changes in recent decades discussed above and that can be summarized in just a word, 'efficiency'. It calls for a dramatic change in current approaches, a process which some countries have already started.

One of the key issues that make the difference in water efficiency is the price paid by the users. The significant differences in the price paid for 200 m^3 of tap water (excluding sanitation) around the world are quite astounding (IWA, 2010). The wide divergence between some of the world's largest cities is hard to understand. The highest price is US \$765 in Copenhagen while the lowest, in Milan at US \$33, is not even 5% of the Copenhagen rate, a disparity that is explained qualitatively by historical reasons (Bru and Cabrera, 2010) and quantitatively by cost structures (Cabrera *et. al.*, 2011). Although elements of the lower bills are subsidised and some costs, especially environmental ones, are ignored, the differences will continue to exist. However, because water is of great social importance, decision-makers are not prepared to break with ancient customs (Frederiksen, 2007).

In the light of the foregoing, a natural transition to a low carbon urban water cycle involves educating both decision-makers and the public. The former will learn how much is at stake and will not hesitate to drive change, while the latter will bear the sacrifices they are asked to make because they understand the need for the decisions. This will be a gradual change based on the following five points:

1. A cross-cutting approach and breadth of vision. Today the water issue is part of a multidimensional space ("Hydraulic engineers have to think bigger and broader"; ASCE, 1996) which should at least consider the three key factors in sustainability. Nor can the problems be solved without breadth of vision ("Think regionally, act locally"; Falkenmark, 1998).

2. Integration into operations. Water policies influence and are influenced by other policies. As noted in the introductory passages of this report, the relationships between water and land use as well as between water and energy use are increasingly recognised as being key areas for policy (and policy tool) integration. Such a cross-cutting approach is especially significant in the context of urban water use.

3. Joining science and politics. At present there is a lack of common ground in all matters relating to the environment (Gleick *et. al.*, 2010) because whilst science follows the pace of events in real time, politics usually responds with a shortened

time horizon due to the nature of political cycles. And while the unanimous recommendations of experts, supported by facts, continue to be ignored, policies will do no more than remedy faults and, in the long term, will be much more expensive. The old adage that "prevention is better than cure" cannot be disputed.

4. Adapting government to the actual context, as only it can be the driving force behind progressive change (Frederiksen, 2007). Coordinated governance is required with clear ideas about how to *integrate* policies. Typically, authority over water resources is fragmented which makes coordinating actions problematic. This is a difficult issue especially in countries which are reactive to change, and is one of the major obstacles that has to be overcome to solve the water problem in Mexico City (Delgado et al., 2006). However, it is not easy to reform very old institutions and laws that protect both vested interests and rights acquired a long time ago. Yet there are some countries which are undertaking such changes due to the evident need for them (Saleth and Dinar, 2005).

5. Enhancing demand management to mitigate water stress. This entails improving efficiency, encouraging saving and promoting reuse. With growing demand for water in a probable scenario of climate change, and irrespective of any further increases in supply wherever these can reasonably be made, the future unavoidably involves more efficient water use

5.8. Training a new generation of decision-makers

The problems to be tackled are formidable and Mexico City offers a catalogue of the most significant. After relentless population growth it now has over 20 million inhabitants. The city's water used to come mostly from its aquifers because the surface resources in the surroundings are scarce. However, the city's unremitting growth and the degradation of its aquifers, which have been severely affected by urban activity, made it necessary to find resources in neighbouring basins, and this generated serious social problems for the farmers who until then had been using this water (Tortajada, 2006; Perlo and González, 2009). And then of course there is the high level of leakage that can only be remedied by huge investment, which will in turn require raising prices with all the social problems that this generates for the poorest classes. The energy implications of this set of circumstances are clearly significant and Mexico City provides a keen example of how one-dimensional engineering problems (Reed and Kasprzyk, 2009) become multidimensional. Technology, although it will always play an important role, is today taking a backseat.

The Mexico City example illustrates how profound changes have taken place over recent decades in the nature and complexity of the challenges which face our community representatives and decision takers. Future water policies need to be more integrated, especially in urban areas where the social implications are greatest. This calls for a cultural change, particularly in terms of the major players, ranging from decision-makers to researchers enamoured with their ivory towers and unlikely to venture out into a broader world because current scientific assessment systems do not reward it. It is in fact easier to achieve scientific excellence by going deeper into a specific field than by trying to delve into the complexity of current problems (Keil, 2009).

In short, there is a need to map out new curricula (Briscoe, 2010) which both bring the training of those who will be the leaders of the future into closer contact with reality and also take into account the three areas (political, managerial and technical) in which decisions are made. This training, which will prepare these future leaders for the role they are to play and will instil a common knowledge of the issues tackled in order to understand the viewpoint of the other positions. Only through integration can sustainable solutions be found.

Yet training for those who make the decisions is not enough on its own. Citizens need to be informed about the environment as there will always be a conflict between what is good in the short term and what is good in the medium-to-long term. This is because while politicians make their decisions with an eye on the next election, solutions are sustainable only if they take into account the interests of future generations. Yet political interests and sustainability can be made compatible by educating the public. If most voters support an unpopular decision because they understand the need for it, their political representatives will adopt it even though the results will only become visible in the medium-to-long term.

The need to deal with problems using an interdisciplinary approach is obvious, and perhaps as a result it has become increasingly prominent. Experts are now beginning to take the idea on board and it is already present in international forums. Here the remark made by Professor Stephenson, a civil engineer, about the World Water Forum in Kyoto in 2003 which he attended representing the IAHR (IAHR, 2003) is very relevant: "*It enabled us hydraulic engineers to realise that we are only a drop in the ocean.*" This interdisciplinary approach has not yet been put into practice, and this is one of the major outstanding issues. Hence, unless the training of decision-makers and society is adapted to the new scenario, it will be virtually impossible to ensure in the near future that most of the people living on the planet (hopefully all of them) will be able to drink quality water at a reasonable price in their homes. Natural resources will be managed in a very efficient way, just if all the actors involved in the problem go in the same direction.

6. CONCLUSIONS

Addressing the supply of quality water to a growing population in a world that is changing at breakneck speed is one of the biggest challenges facing society in the 21st century. Attempting at the same time to reduce the amount of embedded carbon and energy used in the delivery systems provides an added test. So far, the most frequent answer to this formidable problem set has been to continue doing what has always been done, when in fact ensuring a reasonably optimistic future for the generations to come calls for far-reaching changes. Or to put it another way, in a world that is rapidly changing, the response time of solutions has to be similar. Furthermore, it should be borne in mind that challenges are also opportunities, because contemporary globalisation makes it possible not only to share problems, including economic ones, but also to maximise the benefits of knowledge development and the export of technology. This is a new scenario in which efficiency must play a crucial, a unique role, far from what has been up to now in water policies all around the world. A new way that requires cultural change. Our responsibility is to make evident this need.

The water supply and wastewater industries throughout Europe are significant consumers of energy and emitters of greenhouse gases. Wastewater for the EU27 countries accounts for about 0.46% of total emissions and for water treatment the value is similar. Although this is a small percentage of total GHG emissions (the energy, manufacturing and transport sectors are by far the largest emission sources) post-delivery energy use for (e.g.) water heating is thought to increase the level significantly. Within EU and national commitments, they need to make emissions reductions of 20% (26% in the UK) by 2020 and 80% by 2050. At the same time, there will potentially be decreasing water resources in some regions due to the effects of climate change.

Demand reduction, through the reduction of waste and increasing water efficiency at the point of use should be part of the solution to both problems. Savings of up to 10% should be achievable by 2020 and at least 20% by 2050. These will have a proportional effect on energy and emissions for the water supply, but a much smaller effect on wastewater treatment, where they will only reduce the relatively small pumping requirement. Construction and decommissioning were fairly small components of the total carbon footprint (5–15%) due to the long lifetime of capital assets.

Progressive improvements in the efficiency of conveyance and distribution should be able to reduce these components, which are about 40% of the total energy requirement for supply, by 30–50%. Long distance conveyance and distribution is energy intensive: the emissions can exceed those from treatment of lower quality water. It should be avoided in favour of local sources wherever possible.

Improvements to motor and pumping efficiency within water treatment works could reduce this use of energy by 10–45%. Other **operational efficiency** improvements have been shown to reduce the total energy consumption by up to 40% in some cases. Replacement of GHG-intensive treatment processes, such as GAC filtration, has also saved up to 40% of emissions. These savings may not necessarily all be combined, but short to medium term reductions of 20% and longer term reductions of 40% or more appear feasible.

The developments found for the water supply sector probably fall short of the 80% emission reduction target for 2050. This is especially true if source restrictions force the use of lower-quality supplies, such as recycled, brackish or desalinated water. The energy cost of treating these is falling, but is still generally higher than freshwater. A possible solution is to supply water for landscape irrigation, agriculture, toilet flushing in office buildings, cooling towers and industrial processes from such sources, treated to a lower standard, to ensure that freshwater is available to be treated to potable standards. This would reduce the pressure on freshwater and reduce the emissions intensity of treating low quality sources.

In wastewater treatment, improvements to the efficiency of pumps and other motors could provide efficiency gains of 10–40% in this aspect, although it is a smaller proportion of the total than it is in water supply. Other general operational improvements may be capable of saving up to 25% of total plant energy.

Aerobic treatments, especially the activated sludge process, are major consumers of energy and the focus for more substantial reductions. Better process control and other efficiency measures, some of which might be combined, have been shown to reduce consumption by 10–50%. These should enable reductions of 20% of total emissions in the short to medium term.

In the longer term (2030–2050), more substantial reductions in energy consumption and GHG emissions are likely to require the replacement of the main aerobic treatment processes by anaerobic treatments. These are more energy-efficient and can also produce usable biogas.

Biogas from anaerobic digestion of sewage sludge is an important source of renewable energy and should be maximized by ensuring that it is captured and used for combined heat and power generation. It has already been demonstrated in practice that it is possible to make wastewater treatment plants energy neutral by this method. A move to anaerobic primary treatment should reduce the plant energy requirements and increase biogas production, making them into net energy exporters. This will help to offset the remaining energy use by the supply sector.

Dewatered or dried sludge can also be used as a fuel. Sending sludge to landfill should be avoided, because it has high direct GHG emissions and provides no benefits. Agricultural use is preferable, because it can displace some chemical fertilisers, but still results in high emissions. The preferred option should be used as fuel by co-firing or incineration with heat recovery, so that the energy is utilised.

There is a growing interest in integrated systems providing local water treatment and source separation of wastewater with recycling of treated water into non-potable supplies. These may also include local sources, such as rainwater harvesting and surface water. While these approaches can clearly reduce the demand for extraction from fresh surface water and groundwater sources, it cannot be assumed that they will necessarily reduce greenhouse gas emissions. There have been very few life cycle assessments of these, though some indicate that they will have lower emissions than current centralised systems, but possibly not compared with systems optimised for energy production and greenhouse gas minimisation. This is an active area where research is still required.

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The research leading to these results has received funding from the European Union Seventh Framework Programme (FP7/2007-2013) under grant agreement n° 265122.