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WASTEWATER IRRIGATION: THE STATE OF PLAY

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ABSTRACT

As competition among industrial, agricultural, urban and environmental sectors for freshwater intensifies, wastewater is frequently being seen as a valuable resource rather than mere waste. Furthermore, wise reuse of this water alleviates environmental concerns attendant with its discharge to coastal environments and inland waterways. Globally, around 20 million ha of land are irrigated with wastewater, either neat or partially diluted. This figure is likely to increase markedly over the next few decades in response to rising levels of water stress in inhabited catchments – in 1995 around 2.3 billion people lived in river basins considered to be water stressed and this number is expected to increase to 3.5 billion by 2025. Here we review the current status of wastewater irrigation by providing an overview of the extent of the practice in different parts of the world and through an assessment of the current understanding of various issues relating to sustainable and safe management of irrigation with wastewater. A theme that emerges is that wastewater irrigation is not only more common in water stressed regions such as the Near East, but the rationale for the practice also tends to differ between the developing and developed worlds. In developing nations the prime drivers for wastewater irrigation appears to be livelihood dependence and food security, whereas environmental agenda appear to hold greater sway in the developed world. The following were identified as key areas requiring greater understanding for the long-term sustainability of wastewater irrigation: (i) accumulation of bio-available forms of heavy metals in soils, (ii) an understanding of the balance of various factors affecting the environmental fate of organics in wastewater irrigated soils (iii) the influence of reuse schemes on catchment hydrology, including transport of salt loads, (iv) risk models for helminth infections (pertinent to developing nations), (v) microbiological contamination risks for aquifers and surface waters, (vi) transfer efficiencies of chemical contaminants from soil to plant, (vii) effects of chronic exposure of people to chemical contaminants in wastewater, and (ix) strategies for engaging the public in wastewater irrigation schemes.

Key-Words
Human health, irrigation, reclaimed water, recycled water, wastewater, review, risk analysis, salinity, sodicity.
INTRODUCTION

In 1995 about 2.3 billion people (41% of the world’s population at the time) resided in river basins considered to be water stressed (< 1,700 m$^3$ person$^{-1}$ yr$^{-1}$) and this value was predicted to increase to 3.5 billion by 2025 (48% of the projected population) (Water Resources Institute, 2000). Many strategies will need to be implemented over the coming decades to deal with water stress, and wastewater irrigation will undoubtedly be an important part of the mix. Wastewaters of municipal and industrial origin are used to irrigate a wide variety of crops and landscapes across the world. There are many drivers for wastewater irrigation, but they can broadly be divided into societal well-being and environmental protection. The former includes such considerations as livelihood dependence of farmers (increased production/profits through continuity of water supply and, in some cases, the nutritive value of wastewater), and increased food supply and subsequent lessening of malnutrition in poor communities. These in turn contribute to the ultimate benefit of improved standards of living and social welfare. The environmental drivers are two-fold (Anderson, 2003; Hamilton et al., 2006c). First, through substitution of wastewater for conventional irrigation water, environmental impacts associated with heavy abstraction of natural surface-waters and ground-waters are mitigated. Second, reclamation and reuse leads to reduced discharge of wastewater to the environment, particularly to sensitive coastal, lacustrine and riverine systems.

While wastewater irrigation can help realize societal and environmental goals, it can simultaneously be detrimental to these objectives. The potential for transmission of devastating pathogenic diseases is real, particularly in developing countries, where poorly-treated or untreated wastewater is often used for irrigation. Degradation of agricultural soils and contamination of aquifers and surface waters are also significant threats. The reality of challenges facing wastewater irrigation are perhaps best summarized by Part 1 of the Hyderabad Declaration on Wastewater Use in Agriculture, in which water, health, environmental and agricultural professionals from 27 institutions across 18 countries formally recognized that:

- wastewater (raw, diluted or treated) is a resource of increasing global importance, particularly in urban 27 and peri-urban agriculture; 28
with proper management, wastewater use contributes significantly to sustaining livelihoods, food security and the quality of the environment; 2
without proper management, wastewater use poses serious risks to human health and the environment. 

Wastewater irrigation is relevant to major global development objectives, most notably Target 10 of the United Nation’s Millennium Development Goal 7, which sets the challenge of halving the number of people living without access to water supplies or effective sanitation by 2015. If attained, this will see an additional 1.6 billion people (1/4 of the world’s current population) with ready access to a water supply (United Nations, 2000). This will have major implications for wastewater irrigation. Improved sanitation infrastructure (particularly centralized sewerage systems) will lead to greater volumes of available wastewater. The current extent of sanitation, particularly sewerage systems, is markedly lower in the developing world; yet it is also worth noting that even in Europe about one third of the wastewater collected by sewerage systems does not undergo treatment (Table 2). Furthermore, increased access to surface waters and aquifers for potable use will in many cases see lower quality water, mostly wastewater, used for agricultural irrigation.

Possible location for Table 2

This paper outlines the current status of wastewater irrigation. It starts with descriptions of the extent of the practice in different parts of the world and then progresses to discussions on management issues pertaining to wastewater irrigation and attendant agricultural, environmental, and human benefits and risks. Knowledge gaps are identified with the intent of encouraging further research to facilitate sustainable and safe wastewater irrigation.

WASTEWATER IRRIGATION AROUND THE WORLD

Complications of definition and lack of data make it difficult to arrive at a robust figure for global wastewater reuse for irrigation. Nonetheless, attempts have been made, and probably the most well known is 2001 estimate that globally 20 million ha of land was irrigated with wastewater, neat or partially diluted (Future Harvest, 2001). Some qualifications and observations on this estimate are discussed by Scott et al. (2004). The estimate was derived from a combination of statistics and assumptions on sewage generated, treatment capacities, the proportion of peri-urban areas without wastewater demand for agriculture, mixing ratios with conventional surface waters, and annual irrigation depths. By not distinguishing between diluted and undiluted wastewater irrigation it does not give us a handle on the extent of raw wastewater irrigation, which is the practice of key public health concern in the developing world. Lunven (cited in Smit and Nasr, 1992) looked at reuse for agriculture from the viewpoint of the consuming population, and estimated that about 10% of the world’s people consume food produced on wastewater-irrigated land in the early 1990s. This figure is likely to have increased markedly over the last decade. Recently, Bixio et al. (2005a) conducted a comprehensive inventory of reuse schemes on most continents, with the notable exception of Asia, save Japan. To date, a preliminary synthesis of the survey has been presented, but more detailed analyses are forthcoming (Bixio et al., 2005a). The survey reports the volume of wastewater reused for different purposes, but it is restricted to reuse projects (i.e. formal reuse associated with specific sewage treatment plants, STPs) rather than informal reuse practices common in many developing nations, such as irrigation with untreated wastewater. In total, over 3,300 formal reuse projects were identified, with > 800 and > 1,800 being located in the USA and Japan, respectively. This reflects the major disparity in the level of wastewater treatment between developed and developing nations.
In short, we have a poor understanding of the prevalence of wastewater irrigation across the globe. To redress this, van der Hoek (2004) has proposed a global database for the assessment of wastewater use in agriculture. The purpose extends well beyond obtaining a global estimate of the amount of wastewater reuse. The database could be interrogated and models constructed to determine the likely impact of current and potential wastewater practices on agriculture, economies and livelihoods. A key feature of the proposed database is the use of a common typology, the lack of which has stifled previous attempts to obtain comparable estimates of reuse. Irrespective of the true extent of wastewater irrigation, few would debate its increasing significance to agriculture, and there are many significant examples of wastewater irrigation across the globe (Table 2). Moreover, placing aside limitations of definition, reuse statistics are available for many countries (mostly total or proportionate volume, or area of land irrigated), and these give us some appreciation of regional trends. Below is a brief roundup of wastewater irrigation throughout the world.
The Near East

The use of wastewater for irrigation is particularly widespread in the Near East, undoubtedly as a consequence of severe water scarcity coupled with large populations. About a quarter of Kuwait’s agriculture, including amenity horticulture, is irrigated with reclaimed wastewater, and in Iran about 70 million m$^3$ y$^{-1}$ of wastewater is used for agricultural irrigation (Radcliffe, 2004). Oman only treats about half of its wastewater, but over 90% of this is reused, mostly for arboriculture (FAO, 2001).

Israel is a world leader in wastewater irrigation. Israel reclaims > 60% of its sewage effluent, with the overwhelming use being agricultural irrigation (Lawhon and Schwartz, 2006). Historically, agriculture in Israel has been heavily reliant upon two large aquifers, but an increased population size coupled with higher standards of living led to an unsustainable gap between supply and demand (Oron, 1998). One response was the construction of the National Water Carrier, which supplies southern Israel with water from the Sea of Galilee (Kinneret Lake). The other major strategy was to turn to reclaimed wastewater for irrigated agriculture, and there are at present five large-scale reuse schemes (USEPA and USAID, 2004). The earliest, and still the largest, is the Dan Region Reclamation Scheme near Tel Aviv (Kanarek and Michali, 1996). It involves storage of treated effluent from the Soreq STP in an aquifer, and subsequent recovery and distribution to irrigation networks in the southern coastal plain and the Negev district in the north. The next largest scheme, the Kishon project, is a vastly different operation. A 12-million-m$^3$ irrigation storage reservoir located 30 km east of Haifa is supplied with treated wastewater from the Haifa STP and with local wastewater and stormwater, and this is used to irrigate about 15,000 ha of non-edible crops, mostly cotton. A notable feature of the Israeli approach to wastewater irrigation has been the extensive use of drip irrigation (particularly sub-surface) to minimize crop contamination and water loss (Oron, 1998). If Israel reaches its ambitious target of treating and reusing nearly all of its wastewater by 2010 (400 million m$^3$ y$^{-1}$), then 20% of all freshwater sequestered will be reclaimed wastewater (USEPA and USAID, 2004).
Reclaimed wastewater accounts for about 10% (74 million m$^3$ y$^{-1}$) of Jordan’s freshwater supply (McCornick, 2001). McCornick et al. (2004) identified three types of reuse in Jordan. First, there is limited reuse for irrigation in the immediate vicinity of STPs. Until recently this has largely comprised pilot and demonstration schemes. Second, reuse via wadis – dry creek beds and their associated valleys – is common. Deprived of their natural base flow through over-extraction of groundwater resources in the highlands, the effluent discharged to wadis proves a valuable resource for many farmers. Despite concern from the Ministry of Health about the quality of the water, the dependence of farmers on this resource in certain regions is acknowledged and irrigation of vegetables continues. Third, and most important, is indirect reuse via the King Talal Reservoir scheme. About 80% of Jordan’s treated wastewater, effluent from the Samara STP, is discharged to the Wadi Zarqua and subsequently stored in King Talal Reservoir, from where it is conveyed through irrigation channels to supply agriculture in the southern Jordan Valley (McCornick et al., 2001). This supply is augmented with surface flows into the Wadi Zarqua and direct input into the reservoir from the Abdullah Canal – when flows are sufficient. Legally, downstream of the King Talal Reservoir the water is no longer considered reclaimed, but realistically the quality depends on the proportionate contributions of the different sources and the operating efficiency of the Samara STP.

Saudi Arabia’s water challenges and policies are particularly interesting as they are to a large degree a legacy of its geology. Historically, Saudi Arabia has relied heavily on fossil groundwater (FAO, 2003). In contrast to conventional aquifers, fossil groundwater aquifers are not recharged and are thus a non-renewable resource. The major response to this declining reserve, and burgeoning populations and industry, was to commission seawater desalination schemes on a grand scale; and today these account for about 33% of the total industrial and 38% of the total domestic demands, and make Saudi Arabia the world leader in desalination (FAO, 2003). The use of such an energy-hungry technology has nonetheless only been possible through reliance on another non-renewable geological resource: fossil fuel. Consequently, wastewater reuse has been promoted by the Government in recent years. In 2000 the Riyadh Region Water and Sewerage Authority initiated a scheme whereby 415,000 m$^3$ d$^{-1}$ of tertiary treated disinfected effluent is provided free of charge to industry through a reticulated mains system, but only about 45% is procured, mostly for agriculture (USEPA and USAID, 2004).
North Africa

Tunisia is arguably the pioneer of wastewater irrigation in North Africa (Shetty, 2004). With around 70% and 20% of the urban and rural populations being connected to sewers, it has a relatively mature sewerage infrastructure for a developing nation and this has enabled the implementation of large-scale planned reuse (Ministry of Agriculture, 1998). Tunisia’s first wastewater irrigation project, the Soukra scheme, was developed in the early 1960s (Bahri, 1998). It involved irrigating 1,200 ha (later reduced to 600 ha due to urban encroachment) of citrus orchards with treated effluent from the Charguia STP. Despite this early bold step, irrigation with wastewater remained a rare exception until the severe drought of 1989, when Decree 89-1047 was passed to provide an institutional framework for the management of wastewater for irrigation and to set biological and physico-chemical water quality specifications. This paved the way for more expansive wastewater irrigation. By 1998, 8.74 million m³ of treated effluent was used to irrigate 6,997 ha of land (90% agriculture, 8% golf courses) (Ministry of Agriculture, 1998). Tunisia also has ambitious plans for agricultural reuse in the future, with a vision of irrigating about 20,000–30,000 ha with around 290 million m³ of treated wastewater by 2020 (Ministry of Agriculture, 1998).

Egypt and Morocco are other major reuse practitioners in North Africa (USEPA and USAID, 2004). Egypt irrigates about 42,000 ha of land with treated wastewater, neat or diluted. In the interest of public health protection, Egyptian law relating to wastewater irrigation is notably strict. No wastewater, despite the level of treatment, can be used to irrigate vegetables that are eaten raw. Irrigation of non-food crops and highly protected food crops (e.g. fruit trees) is encouraged. In Morocco, about 8,000 ha of farmland are irrigated with poorly treated or untreated wastewater. The poor quality of this water is probably responsible for the high levels of helminth infections observed in parts of Morocco (Habbari et al., 1999). Perhaps the biggest obstacle to safe and efficient reuse is the aging sewerage infrastructure: around 70% of Morocco’s STPs do not operate to specification.
Sub-Saharan Africa

The extent of wastewater irrigation in Sub-Saharan Africa is unclear. In many countries weak economies, poor institutional structure and limited or dilapidated public assets (sewage and irrigation networks) preclude the development of formal reuse schemes. A survey of farmers in and around Nairobi in Kenya found that 34% appropriated raw sewage from trunk sewers (Hide and Kimani, 2000; Hide et al., 2001). A parallel survey in Kumasi, Ghana, found that 38% of farmers extracted irrigation water from rivers or streams, most of which are heavily polluted with raw sewage (Cornish and Aidoo, 2000; Cornish et al., 2001). Servicing less than 5% of households, Kumasi’s reticulated sewerage system is very small, and most sewage is either discharged directly into rivers and streams or is collected from septic tanks and then disposed of to waterways (Keriata and Drechsel, 2004). Cornish and Kielen (2004) have suggested that scenarios similar to those in Nairobi and Kumasi, i.e. the widespread use of diluted or undiluted untreated wastewater, are probably common throughout much of Sub-Saharan Africa. Indeed, tapping into sewers to procure water for irrigation of vegetable crops (lettuces, tomatoes, onions and eggplants) is known to occur in Dakar, Senegal, just south of the Sahara (Faruqui et al., 2004).

Unlike most of Sub-Saharan Africa, South Africa has a well developed sewerage infrastructure, comprising in excess of 1,000 STPs (Grobicki, 2000). Despite this, less than 3% (41 million m$^3$ y$^{-1}$) of the treated effluent is reused. Aquifer recharge and heavy industry are the dominant applications, although there are some agricultural reuse schemes, such as the irrigation of 22,000 ha with treated effluent from the Johannesburg Northern Works STP (USEPA and USAID, 2004). Zimbabwe also has several treated wastewater irrigation schemes, the two largest being at Harare and Bulawayo (Hranova, 2000).

Asia

Wastewater reuse is well-established in parts of Asia. There is a particularly strong incentive for widespread wastewater reuse in China: it is encumbered with supporting 22% of the World’s population (1.3 billion) with only 8% of its available freshwater resources (Worldwatch Institute, 2006). The problem is exacerbated by the uneven distribution of freshwater, both spatially and temporally, with at least 80% of the total freshwater resources located in south-eastern China (a region that only accounts for 35% of the country’s arable land) and 60% of the precipitation in this region occurring from April to July (Wang, 1999; Deng et al., 2006). Combined, these factors contribute to a per capita water resource availability that is less than a third of the world average (Wang and Jin, 2006). While the extent of wastewater irrigation in China is unknown, it is likely to be considerable. Two decades ago, Bartone and Arlosoroff (1987) suggested a figure of 1.33 million ha of wastewater-irrigated land. A linear programming optimization model constructed by Chu et al. (2004) suggests that annually China could potentially reuse 1.78 billion m$^3$ of its wastewater per year, which represents only 6.3% of the total urban wastewater produced. The model, which accounts for technological, physical and economic constraints, suggests that greater reuse would be hindered by the high costs of reuse schemes and the currently low water prices. Beijing emerged as the province with greatest reuse potential, with the model suggesting that 0.31 billion m$^3$ y$^{-1}$ could be reclaimed, which contrasts starkly with the current level of only 0.04 billion m$^3$ y$^{-1}$. In addition to Beijing, reuse potentials greater than 0.14 billion m$^3$ y$^{-1}$ were calculated for Liaoning, Guandong, and Jiangsu provinces. It is worth noting that when Chu et al. (2004) extended their model to account for uncertainty in parameters, the estimated mean wastewater reuse potential for the nation increased to 2.40 billion m$^3$ y$^{-1}$ (SD 2.75 billion m$^3$). Agriculture emerged as the sector expressing the greatest demand for wastewater (93.8% of total demand) and was also determined to offer the highest reuse potential under the scenario of unchanged agricultural water price (63% and 72% of total reuse potential for models with and without uncertainty, respectively).
Wastewater currently used for irrigation in China is mostly untreated and of poor quality. A survey in 1994 found that about 85% the wastewater used for irrigation did not meet the nation’s standards for re-use (He et al., 2001). Japan is the world’s leader in urban wastewater reuse. While agriculture accounts for only about 13% (20 million m$^3$) of the total wastewater reuse, many of the urban applications also involve irrigation (e.g. of parks, golf courses and sporting fields) (Ogoshi et al., 2001; JSWA, 2002).

Owing to high annual precipitation, wastewater irrigation is uncommon in developing countries in tropical Asia – India and Vietnam being notable exceptions (Radcliffe, 2004). At least 73,000 ha of land in India were irrigated with wastewater over 15 years ago (Strauss and Blumenthal, 1990), and this figure is only likely to have increased and may have been a substantial under-estimate at the time in any case (van der Hoek, 2004). Today, around 40,000 ha of farmland surrounding Hyderabad are irrigated via a large indirect reuse scheme whereby untreated effluent from the city is discharged to the Musi River, which feeds an irrigation network downstream. The arid environment of West Asia is also host to significant reuse. In Pakistan, 50 of the 60 cities with populations over 10,000 use untreated wastewater to irrigate agricultural land (about 32,500 ha country-wide) (Ensink et al., 2004).
Central and South America

Despite the plentiful water resources in much of Central and, particularly, South America, wastewater reuse is receiving moderate and increasing attention. The reason is two-fold. First, the location and size of many mega-cities means that freshwater may not be as abundant or accessible as one might expect. Brazil, for example, hosts about 6% of World’s freshwater, yet 80% is in the Amazon basin in the north, whereas 65% of the population inhabits the southeastern, southern and central-western regions (USEPA and USAID, 2004). These regions are still far from arid, but coupled with escalating populations in already huge metropolises like the São Paulo Metropolitan Region (18 million people), conservation of water resources in cities and their environs is demanding attention (Jacobi, 1997). To date, reuse in Brazil has mostly involved industrial applications, such as for cooling towers at the Mercedes Benz Sao Bernardo do Campo plant, and the under-construction reuse scheme at São Paulo International Airport, which will eventually supply 31% of the airport’s water needs through on-site wastewater recycling (Wagner, 2006). Reuse for agriculture is on the horizon, with feasibility studies and business cases concerning the sale of wastewater to farmers under development (Wagner, 2006).

Wastewater reuse is practiced in the arid coastal zone of Peru, where mostly untreated effluent is used to irrigate a variety of agricultural crops, including, vegetables and many non-food crops (e.g. fodder, arboriculture and cotton) (USEPA and USAID, 2004). The largest scheme is at Lima, where 5,000 ha of agricultural land are irrigated with untreated wastewater. Reuse for agricultural irrigation is practiced in the peri-urban zone of three cities in Bolivia, namely, Cochabamba, La Paz El Alto and Tarija (Durán et al., 2003). Cochabamba involves direct and indirect reuse, whereas only indirect reuse occurs at the other two cities. Salt-related problems associated with poorly treated effluent in Cochabamba have seen many farmers switch from vegetable production to more salt-tolerant fodder crops (Huibers et al., 2004).
The second driver for reuse in South America is pollution mitigation. For example, in an effort to reduce environmental impacts associated with discharge, about 70–80% of Santiago’s raw sewage is used to irrigate most of city’s salad vegetables and low-growing fruits (USEPA and USAID, 2004). Unfortunately, this had a marked detrimental impact on public health, and consequently improved treatment trains and irrigation practices have been implemented.

North America

Mexico irrigates in excess of 350,000 ha of farmland across more than 40 irrigation districts with wastewater, yet only 11% of this wastewater is treated (Peasey et al., 2000). In what is one of the World’s largest and oldest extant large-scale reuse schemes, wastewater from Mexico City is transported about 65 km north to Mezquital, where it feeds an extensive irrigation network that services around 90,000 ha of various crops (vegetables, cereals and fodder) (van der Hoek, 2004).

California and Florida account for the vast majority of wastewater irrigation practiced in the USA. California is the undisputed pioneer, with reuse for agricultural and landscape purposes dating as far back as 1890 and 1912, respectively (RWTF, 2003). By as early as 1970 around 216 million m\(^3\) y\(^{-1}\) of wastewater was reused, and this has increased to an estimated 548–730 million m\(^3\) y\(^{-1}\) (RWTF, 2003). While much of the early reuse was for groundwater recharge, today the agricultural sector is the largest user of reclaimed water (Table 3). In contrast, a similar total volume is used in Florida (803 million m\(^3\) y\(^{-1}\), FDEP, 2002) but the dominant use is landscape irrigation (Table 2). In both California and Florida reclaimed wastewater is highly treated to comply with strict quality standards (DHS, 2001; RCC and WCIWRWG, 2003).

Possible location for Table 3

Owing to a high availability of freshwater in most regions, little wastewater reuse occurs in Canada. Nevertheless, environmental impacts attendant with the disposal of wastewater and economic advantages of convenient distribution in some areas have put reuse on the national agenda in recent years (Marsalek et al., 2002).
Europe

On cursory inspection Europe would not seem a dry continent, but according to the water stress index (WSI) – the ratio of a country’s total water withdrawal to its total renewable freshwater resources – about half of the countries of Europe are in a situation where water availability is becoming a constraint on development and significant investment is required to secure sufficient water supplies (i.e. WSI > 10%) (Bixio et al., 2005b). Such supply-demand imbalances arise from temporal and spatial inconsistencies in rainfall, and large population sizes coupled with relatively high standards of living in many countries. A recent inventory of wastewater reclamation in Europe identified in excess of 200 operating reuse projects and many in the planning stage (Bixio et al., 2005b). Overall, reuse was most common along the coastal region of typically drier southern Europe, and in the heavily-populated urbanized parts of the wetter north (northern continental Europe and England). The types of reuse differed between these two broad groupings, with agricultural irrigation and urban/environmental applications respectively accounting for 44% and 37% of the projects in the south, and the industrial and urban/environmental sectors the dominant users in the north (51% and 33%, respectively).

Comprehensive reviews of wastewater reuse in many European countries have been conducted by Angelakis and Bontoux (2001) and Angelakis et al. (2003), from which the following précis is largely drawn. France has around 20 to 30 water reuse projects and irrigates in excess of 3,000 ha of farmland with mostly treated wastewater. The water is used for a wide variety of purposes, including, vegetables, orchards, cereals, tree plantations and forests, grasslands, public gardens, maize, and golf courses. With a WSI over 20%, Italy falls into the highest water stress category, which calls for ‘comprehensive management efforts to balance supply and demand, and actions to resolve conflicts among competing uses’ (Bixio et al., 2005b). Italy irrigates over 4,000 ha of agricultural land with treated wastewater, and in 2001 Barbagallo et al. (2001) identified 16 new reuse scheme proposals. Spain and Belgium also have very high WSIs (~29% and 42%, respectively), but whereas Spain uses treated wastewater to irrigate golf courses and agriculture, and to recharge aquifers, there is little reuse (mostly industrial) in Belgium, which only treats about 40% of its sewage. An impressive reuse scheme at Vitoria in the Basque country of northern Spain involves treating wastewater to the Californian reuse standards and spray irrigating 9,500 ha of crop in the summer (USEPA and USAID, 2004). The Californian standards are generally regarded as very strict; in brief, wastewater for food crop irrigation must undergo secondary treatment, filtration and disinfection, and a total coliform concentration of \( \leq 2.2 \times 10^0 \) mL\(^{-1}\) (running 7-day median) must be achieved (DHS 2001). Greece also suffers from water shortage (WSI ~ 12%), and with more than 83% of the treated effluents being produced in regions with a deficient water balance, reclaimed wastewater is being considered as a serious option for redressing the problem. Reuse is in its infancy in Greece, but there are at present several pilot projects in progress. Portugal paints a similar picture (WSI ~ 13%) and has several projects in the pipeline—so to say! On the whole, Sweden, like the rest of Scandinavia, is not under water stress, but it does have dry regions. It has been proactive in water conservation and wastewater reuse, and new environmental legislation limiting nitrogen discharge from STPs is likely to encourage further recycling (USEPA and USAID, 2004). There are at present around 40 wastewater irrigation schemes in the dry regions of the southeast. These schemes involve conveying highly-treated wastewater to large storage reservoirs, where it is held for up to 9 months (minimum of 4) before being distributed for irrigation (Angelakis et al., 2001).
Limited wastewater irrigation takes place in the Netherlands, although restrictions and taxes on aquifer abstraction put in place by the Dutch Government are an incentive for further reuse. Aside from some watering of golf courses, parks and road verges, little wastewater irrigation takes place in the United Kingdom. Most wastewater reuse involves maintaining river flows. Negligible or no wastewater irrigation takes place in Austria, Denmark, Finland, Germany, Ireland and Luxembourg. Indeed, reuse of any form is rare in most of these countries and typically limited to industrial applications or aquifer recharge.

Oceania

Most of the islands of the Pacific experience very wet climates and there often is little incentive, or inadequate infrastructure, for reuse. Being the World’s driest inhabited continent and having well-developed sewerage systems makes Australia a notable exception. It has incentive and infrastructure. Australian STPs produce about 1,824 million m$^3$ y$^{-1}$ of effluent, and about 9.1% (166.5 million m$^3$ y$^{-1}$) of this is reclaimed (Radcliffe, 2003). Australia has at least 584 operating reuse schemes. There are 79 and 229 schemes devoted to industrial and urban applications, respectively, and each of these uses account for 22% of the total volume reclaimed (Radcliffe, 2004; Boland et al., 2005). The agricultural sector has around 270 schemes (with irrigation the overwhelmingly dominant use), making it the largest user of reclaimed water in Australia (54% of all reuse by volume). It is also worth noting that the majority of the ‘urban’ reuse is also for irrigation, particularly of golf courses (55% and 49% of total urban reuse by schemes and volume, respectively) and recreational facilities (49% and 48%, respectively). Environmental applications, including wetland augmentation, account for the remaining reclaimed water use (6 schemes, 2%).
Continued growth of Australia’s high value horticultural industry is contingent upon reliable access to water, and to this end reclaimed wastewater is receiving increasing attention (Hamilton et al., 2005a). Horticultural reuse is a particularly attractive proposition because of the high returns per volume of water and because many horticultural districts are located in close proximity to large STPs on the peri-urban fringe of major cities. The Virginia Pipeline Scheme in South Australia, established in 2000, is Australia’s largest reuse scheme for horticulture (Kracman et al., 2001; Kelly et al., 2003). The Virginia region accounts for about 35% of South Australia’s horticultural production, which equates to about AU$120 million (Kracman et al., 2001). The scheme involves providing highly-treated reclaimed water from the Bolivar STP to about 250 vegetable growers. Bolivar is the major STP servicing the State’s capital, Adelaide, and it is located within the immediate vicinity of the Virginia vegetable growing district. The wastewater undergoes standard primary sedimentation, secondary treatment via biological trickling filters, and tertiary treatment in a dissolved air flotation/filtration plant followed by retention in a disinfection and storage contact reservoir. Currently, about 8 million m$^3$ y$^{-1}$ of this reclaimed water is being used by horticulturalists, but the system can potentially deliver 23 million m$^3$ y$^{-1}$ (Kelly et al., 2003; Radcliffe, 2003). Aquifer storage and recovery (ASR) is also being tested at Virginia and in other parts of the country (Dillon et al., 2001).

**MANAGEMENT ISSUES**
Salinity and Sodicity

The concentration of salts in wastewater streams varies considerably from one system to the next and depends on inputs into the sewer. Sewage with a high industrial input tends to be more saline than straight municipal sewage. With typical salinities ranging from 600–1,700 µS cm⁻¹ (Feigin et al., 1991), wastewaters are more often than not more saline than conventional freshwater sources, and these extra salts can pose a threat to the viability of irrigation schemes. The two prime salt-related issues are the effects of salinity on plant growth and changes in soil structure from sodicity – a high proportion of sodium relative to other cations (especially calcium and magnesium).

Salts can affect plants either through causing osmotic stress or via direct toxicity. High concentrations of salts in the root-zone lead to a decrease in the osmotic potential of the soil-water solution, thus retarding the water uptake rate of the plant. The plant expends considerable energy trying to osmotically adjust, by accumulating ions, and this is typically at the expense of yield (Maas and Nieman, 1978; Maas and Grattan, 1999). Toxicity occurs when salt ions enter the plant and interfere with cellular processes. Sodicity alters the physical structure of the soil – the most notable effect being the dispersion of soil aggregates. Dispersion, in combination with other processes, such as swelling and slaking, can ultimately affect plants through decreasing the permeability of water and air through the soil, water-logging, and impeding root penetration (Fig 2). The effects of such processes on cropping systems are reviewed by Rengasamy (2006).

The ‘sodicity potential’ of water is quantified by the sodium absorption ratio, SAR, which is given as:

\[ \text{SAR} = \frac{\text{Na}^{+}}{\sqrt{\left(\text{Ca}^{2+} + \text{Mg}^{2+}\right)/2}}, \]

where the ion concentrations are in milliequivalents L⁻¹. An SAR greater than 3 is broadly regarded presenting a sodicity risk, especially on heavy soils. The SAR of wastewater can vary considerably, but has been reported to be generally well above 3, often be within the range of 4.5–8.0 (Magesan et al., 1999). A dynamic relationship exists between SAR and salinity of water, with increased likelihood of clay dispersion occurring at high SAR, being offset by clay coagulation due to high concentrations of ions in solution (Sumner, 1993). It follows that soils irrigated with water of high SAR and saline water, may be no more at risk of structural problems than those irrigated with very low salinity and moderately low SAR water (Sumner, 1993). The potential for sodicity-related problems can be addressed more directly through calculating the exchangeable sodium potential (ESP) of the soil, which is given as
where ion concentrations are in milliequivalents 100g⁻¹. The degree of impact of sodium on a soil is dependent upon a combination of factors other than ESP. These may include amount of clay, clay mineralogy, and surface charge characteristics (Halliwell et al., 2001; Leeper and Uren, 1993); electrolyte concentration and pH (Nelson et al., 1999; Rengasamy and Sumner, 1998); organic matter (Nelson et al., 1999); moisture content prior to wetting, and wetting drying cycles (Nelson et al., 1998). As such it is difficult to determine at what ESP sodicity will impact on a soil, thus making the establishment of formal guidelines difficult (Sumner, 1993). Nonetheless, some practical guides do exist and perhaps the most widely used classification scheme that of the United States Department of Agriculture (Table 4).

Possible location for Table 4

In Australia, where salinity and sodicity are profound problems, sands and loams are considered saline if the surface layer comprises > 0.1% NaCl (> 0.2% for clay loams and clays) and are classified as sodic if the top meter has an ESP > 6 (Northcote and Skene, 1972). According to these definitions, 5% and 25% of Australia’s surface area was estimated to be saline and sodic, respectively, in the 1970s (Northcote and Skene, 1972), and presumably these figures have increased. Salt-related issues are serious concerns for wastewater irrigation in Australia. In a major horticultural wastewater irrigation scheme recently commissioned in Werribee southern Australia (~30 km west of Melbourne, Victoria) the threat of salt-related problems was tackled from the outset by setting salinity targets for the irrigation water (Melbourne Water and Southern Rural Water 2004, Barker-Reid et al. 2007). The targets involve many qualifications and are described in detail by Melbourne Water and Southern Rural Water (2004). In brief, short-term targets ranging from 1,400–1,800 µS cm⁻¹ and a longer-term target of 1,000 µS cm⁻¹ were set. The short-term targets are being met through mixing of the fairly saline treated wastewater (annual average ~1,700 µS cm⁻¹) with river water. By 2009 the intent is to have reduced the salinity of the neat treated wastewater to the 1,000 µS cm⁻¹ target, which would mean that mixing with river water would no longer be necessary. This will be achieved through the addition of a desalination plant to the treatment process and via strategies and policies intended to reduce salt inputs into the sewerage system, particularly those industrial from heavy industry. Controlling salt inputs into the sewer has proved a successful strategy in Israel, where strict regulations prohibiting the discharge of brines from various industries into the sewer as well as a labeling standard for domestic detergents have undoubtedly been key elements behind the successful long-term practice of wastewater irrigation (Reid and Sarkis, 2006).
Salt-related issues can also be managed at the farm scale. Considering that most horticultural crops uptake salts more readily through the leaves than through the roots (Maas, 1985), the toxic effects of salinity can be mitigated through substituting over-head irrigation with drip or furrow methods. Realistically though, irrigation practices are often culturally well entrenched and capital investment in new infrastructure may not be feasible for many farmers. Soil structure that has been degraded through sodicity-related processes can often be restored through the application of soil amendments such as calcium carbonate (lime), calcium sulphate (gypsum), animal manure, and plant organic matter (e.g. wheat straw) (Maher et al., 2004). Also, deep tillage can be used to bring calcium-rich sub-soils to the surface to restore soil structure (Loveday and Bridge, 1983). Another approach is to recurrently flush sodium from the soil with low-sodium water and collect the leachate (Surapaneni and Olsson, 2002). Nonetheless, the long-term accumulation of sodium in soils under wastewater irrigation remains a serious issue, and our understanding of the processes of sodicity, particularly in relation to the dispersive effects of exchangeable ions in different soil types, requires further research (Churchman et al., 1993).

In addition to the agricultural risks attendant with wastewater irrigation, there are salt-related environmental threats to be considered for any scheme. Questions relating to the existing salinity of the aquifer, whether or not it discharges to surface waters, and what the other uses of the aquifer are (e.g. drinking water) need to be answered (Bond, 1998).

Nutrients
An attractive property of reclaimed wastewater in contrast to conventional irrigation waters is its potential fertilizing capacity. Yield increases of various crops – celery, eggplant, lettuce, maize and sorghum – have been attributed to irrigation with nutrient-laden wastewater (Kaddous and Stubbs, 1983; Chakrabarti, 1995; Al-Nakshabandi et al., 1997; Sheikh et al., 1998; Marceos Do Monte, 1998). Clearly, an appropriate fertilizer regime will also meet the crop’s needs, but in developing nations the application of fertilizers is often not an economically feasible option, and the supply of free nutrients in the irrigation water is clearly an attractive proposition. Indeed, in Mexico’s Mezquital Valley farmers protested the Government’s proposal to upgrade the level of wastewater treatment for fear of the water losing its fertilizing capability (USEPA and USAID, 2004).

On the other hand, the high concentrations of nutrients in wastewater can be agriculturally and environmentally problematic. Over fertilization through irrigation with reclaimed water has been shown to affect the yield and maturation of perennial crops, or reduce fruit size and quality, through the excessive application of nitrogen (Baier and Fryer, 1973). Delayed maturation of sunflowers following irrigation with high nitrogen (30 mg L⁻¹) wastewater has also been reported (Marceos Do Monte, 1998). A high nutrient loading can also affect the hydraulic conductivity of the soil. Effluents with a high carbon to nitrogen ratio can promote excessive growth of the soil microfauna, which, through clogging pores in the soil matrix, can lead to a reduction in hydraulic conductivity (Magesan et al., 2000).

Over fertilization also has the potential to affect a plant’s ability to resist disease. While this issue has not been well studied with particular reference to wastewater, there are many examples from the fertilizer literature. For example, the application of high doses of nitrogen has been shown to increase the severity of pre or post harvest disease in onions (Wright, 1993), potatoes (Kumar et al., 1991), strawberries and apples (Kolbe, 1977), barley (Jensen and Munk, 1997), rice (Long et al., 2000), cranberry fruit (Davenport and Provost, 1994), and tomato (Hoffland et al., 2000). Over-abundance of nitrogen can lead to an increase in vegetative growth, producing large, fast growing tissues that are structurally weaker and more susceptible to pathogen attack. As pathogens derive their nutritional requirements from the host cells, the abundance of nitrogen and other elements may also directly affect their growth (Solomon et al., 2003). Excess nitrogen has also been shown to decrease the plant’s ability to synthesize compounds important to its natural defenses (Kumar et al., 1991; Matsuyama and Dimond, 1973). The form in which nitrogen is supplied is also important, and can determine if disease susceptibility is increased or decreased in a particular crop (Huber and Watson, 1974). In general, nitrogen supplied as ammonium reduces the uptake of calcium and other cations while nitrate stimulates cation uptake. In some instances, differences in pathogen responses to elevated nitrogen levels in plant tissues have been observed. For example, Hoffland et al. (1999, 2000) found that Pseudomonas syringae and Odium lycopersicum infections increased as nitrogen content increased, Botrytis cinerea decreased, and Fusarium oxysporum did not change.
Leaching of nitrates and other solutes poses one of the greatest threats to groundwater health arising from wastewater irrigation (Bond et al., 1998). The risk of groundwater contamination with nitrate can be markedly reduced through appropriately matching cropping systems to effluent characteristics (Snow et al., 1999). For example, high-yielding crops with large amounts of nitrogen in their biomass would be more effective than tree plantations at reducing nitrate leaching. Nitrate is very soluble and is not easily fixed to soil clay minerals, and hence, if not taken up by plants, is readily leached through drainage (Hermon et al., 2006). Phosphorus is more readily fixed within soils, and therefore its transfer from soils is usually associated with plant uptake or soil transport processes such as erosion. But subsurface pathways such as preferential flow have gained significant attention in recent times (Gupta et al., 1999; Magesan et al., 1999; Simard et al., 2000; Garnet et al., 2004). The phosphate retention capacity of soil has also been implicated as a factor in phosphorus leaching in soils irrigated with effluent (Magesan et al., 2000).

**Heavy Metals and other Inorganic Contaminants**

Raw sewage typically contains significant concentrations of inorganic chemicals, but their concentrations in STP effluents are markedly lower. Most metals, for example, are cationic and are strongly sorbed to negatively charged organic matter and clay minerals, and consequently precipitate out of the sewage to the sludge (biosolid) component in standard sewage treatment processes (Stevens and McLaughlin, 2006). Therefore, most concern over the potential effects of metals on plant production and human health relates to raw sewage irrigation or the use of biosolids as fertilizers rather than irrigation with treated wastewaters. Boron is a notable exception: under pH ranges of typical wastewater streams, it is found in the uncharged boric acid ionic form, and therefore mostly remains in the effluent. Its relatively high concentrations in treated effluents can be toxic to many plants and consequently a significant constraint for many wastewater irrigation schemes (Unkovich et al., 2006).
Few wastewater irrigation schemes have been operating for a sufficient period to enable the long-term accumulation of heavy metals in soils to be studied. An exception is a series of studies conducted on pastures at the Western Treatment Plant (WTP), 35 km West of Melbourne, Victoria, Australia. Some pastures at the WTP have been irrigated with raw sewage for over 100 years, and others for shorter periods of time. A survey of metal concentrations in these soils enabled a picture of metal accumulation over time to be developed (Xiong et al., 2001) (Fig 1). After 107 years of irrigation with raw sewage, cadmium and zinc concentrations in the soil were close to the Australian Ecological Investigation Levels (EILs) for soils (NRMMC et al., 2006). Nevertheless, the phytotoxic effects of metals cannot simply be considered in terms of total concentration: the bioavailability of different fractions needs to be accounted for. A study on heavy metal fractionation at the WTP has shown that a reducible fraction (‘Fraction IV’) is the most abundant form of the metals shown in Fig 1 (Xiong et al., 2004). This fraction is potentially bio-available, as the metals may be released as iron oxides if the redox potential of the soil decreases. But a further study of copper in the WTP soils demonstrated that most of it is strongly sorbed to the soils, and this may explain the relatively low concentrations in plant tissues despite the high soil concentrations (Li et al., 2006).

Possible location for Figure 1.

While the accumulation of metals in the soils will vary from one wastewater-irrigation scheme to the next, as a function of factors such as soil type, crop type and metal concentrations in the irrigation water, the study of Xiong et al. (2001) nonetheless demonstrates the capacity for metals to slowly accumulate over time. It is particularly relevant to situations where untreated wastewater is used, i.e. many developing world scenarios. The potential for metals to confer health problems to consumers of wastewater-irrigated crops needs to consider bioavailability and food chain transfer. Metal mobility and bioavailability in soil varies significantly with soil properties for similar total soil metal concentrations. Some metals pose little hazard through food chain contamination due to their strong phytotoxic effects (i.e. increasing metal concentrations cause plant mortality before transfer to the next trophic level, e.g. people or grazing animals, has an opportunity to occur). This is known as the ‘soil-plant barrier’. Metals can be assigned to four groups based on their retention in soil and translocation within the plant (Table 5). Cadmium has been identified as the major heavy metal of health concern in sewage as it is, relative to most other metals, more available to plants and is found in concentrations in harvestable portions of the crops that could be harmful to humans, but which are not toxic to the plant (Stevens and McLaughlin, 2006).
Environmental Fate of Pesticides and other Organic Chemicals

There are many organic contaminants in wastewater that can pose a threat to human and environmental health (Ying 2006). Specific concerns relating to human health risks are dealt with in a separate section below, but before considering these it is worthwhile addressing wastewater-specific issues relating to the environmental fate of organics, as these can have implications for natural systems, groundwater supplies, agricultural production and food safety. In the context of contaminant transport and transformation, wastewater has certain characteristics – most notably high dissolved organic matter (DOM) – that distinguish it from conventional irrigation waters. It therefore warrants specific attention in this regard; yet, as noted in a recent review, our knowledge of wastewater influences on such processes is poor, and results across studies are inconsistent (Müller et al. 2007). Focusing on pesticides, Müller et al. (2007) noted wastewater irrigation often increases pesticide mobility, through the formation of soluble pesticide-DOM complexes, and enhances pesticide degradation through providing energy for certain microorganisms. On the other hand, reduced pesticides mobility, owing to increased sorption, has been observed. Solvents and surfactants, which are found in considerably higher concentrations in wastewaters than conventional irrigation waters, can increase the solubility and sorption of hydrophobic pesticides and thus ultimately influence mobility. Müller et al. (2007) suggest that the spatial variability of soils and/or variation in the composition of wastewaters (e.g. concentration and properties of DOM, and concentrations of surfactants and solvents) may explain the inconsistent findings across studies of wastewater impacts on contaminant fates.
Wastewater Irrigation in a Catchment and Landscape Context

Management issues relating to wastewater irrigation are often considered in a local, on-site context only. Successful implementation of sustainable wastewater irrigation schemes also requires consideration at the catchment and landscape levels. The role that the irrigation project plays in the hydrological balance of the catchment needs to be understood. Hydrology models that are currently widely used for catchment management could readily be adapted to incorporate wastewater irrigation schemes. A particularly useful application would be to study the transport of salts to aquifers and surface waters. Most wastewaters, particularly those with a high industrial waste component, are markedly more saline than surface freshwater. One means of managing salts on site is to leach them through the soil profile (Rengasamy, 2006), and while there are systems for collecting the salty leachate (Jayawardane et al., 2001), in many situations it simply percolates through to the groundwater. But what happens next? Does the aquifer act as a salt store, or is much of the salt exported to surface waters? As noted above, transport of salts to aquifers and surface waters can cause environmental and agricultural problems. The influence of a wastewater irrigation scheme on salt and hydrological balances in a catchment needs to be considered in any proposal.

Catchment hydrology models comprise a number of water, energy and vegetation processes that are interrelated and distributed over space and time. A typical model consists of a large number of coupled equations describing the direction of water flow, including surface and sub-surface flows, which are integrated to provide predictions of monthly and annual stream-flow (Wagener and Gupta, 2005). These models can be used to quantify surface and groundwater contributions to salt export at a catchment scale. An example is 2CSalt, which has been used in Australia to determine the impact of land-use practices (but not yet wastewater irrigation) on salt and water yield (Weeks et al., 2005; Beverly et al., 2006). The 2CSalt model is a component of the Catchment Analysis Tool (CAT) – a software interface and toolbox that can be used to assess impacts of point-scale land use activities on both paddock and catchment scale processes, such as surface run-off, sediment loss, nitrogen mobilization, biomass yields, stream quality, and groundwater discharge (CAT, 2004). Many catchment models only operate at the catchment level, but those that link paddock and catchment scales could prove useful for including wastewater irrigation schemes in landscape planning.
The importance of geography in a broad sense cannot be understated when evaluating wastewater irrigation schemes. Water is heavy, so pumping it over long distances and/or up-hill can be prohibitively expensive. Costs are not simply those associated directly with fossil fuel consumption, but should also include the environmental externalities, i.e. greenhouse gas emissions. Recently, a wastewater irrigation proposal in Queensland, Australia, was vetoed largely because of the environmental cost attendant with pumping water up-hill (SEQRWTF, 2003). Fortunately, in many parts of the world vegetable market gardens and STPs tend to be located in close proximity, i.e. in the peri-urban zone on the outskirts of cities. In Greece, for example, 88% of its treated effluents are located within 5 km of agricultural land requiring irrigation water (Tchobanoglous and Angelakis, 1996). In South Australia, the treated wastewater from Adelaide’s Bolivar sewage treatment plant is distributed to 250 vegetable growers in the adjacent Virginia Plains district (Kracman et al., 2001; Kelly et al., 2003); and effluents from two large sewage treatment plants on the eastern and western fringes of Melbourne are being used to irrigate nearby market garden districts (DSE, 2003; Arbon and Ireland, 2003).

Pathogens

Considering its origin it is not surprising that municipal wastewater can contain a wide variety of microorganisms that are pathogenic to humans. These include bacteria (e.g. Salmonella spp., Shigella spp. and enteropathogenic Escherichia coli), viruses (e.g. adenovirus, poliovirus, hepatitis A virus and rotavirus), protozoans (e.g. Cryptosporidium parvum, Giardia intestinalis (formerly G. lamblia) and Entamoeba histolytica), and parasitic helminthic worms (e.g. Ascaris lumbricoides, Necator americanus and Trichuris trichiura) (Yates and Gerba, 1997; Toze, 2006). Wastewater can potentially be responsible for several diseases and conditions resulting from infection with these pathogens. These include typhoid (Salmonella spp.), dysentery (Shigella spp. and E. histolytica), gastroenteritis (enteropathogenic E. coli), diarrhoea, vomiting or malabsorption (adenovirus, rotavirus, C. parvum, G. lamblia and T. trichiura), cholera (V. cholera), ascariasis (A. lumbricoides), and anaemia (N. americanus) (Yates and Gerba, 1997). The pathogen profile and concentrations of specific pathogens in raw sewage will depend on epidemiological status of the contributing population (Gerba and Rose, 2003).
Health risks associated with human exposure to pathogens can be quantified in one of two ways: epidemiological studies and a probabilistic modeling technique known as Quantitative Microbial Risk Assessment (QMRA). Both approaches have their pros and cons, and should be seen as complementary rather than mutually exclusive. Epidemiological studies involve relating exposure factors to the incidence – spatially and temporally – of a disease in a population. A comprehensive synthesis and analysis of the many epidemiological studies that have been undertaken on wastewater irrigation was conducted by Blumenthal and Peasey (2002). An important feature of the analysis was that only studies satisfying the following criteria were considered: well-defined exposure and disease, risk estimates calculated after allowance for confounding factors, and statistical associations between exposure and disease. Clearly, each case study and wastewater irrigation scenario is unique, but it is nonetheless important to look for general patterns. Some of the key findings of Blumenthal and Peasey’s analysis can be summarized as follows.

There was evidence to suggest that unrestricted irrigation of vegetable crops with untreated wastewater led to increased incidence of helminth (mostly *A. lumbricoides*) infection, bacterial infections (typhoid, *Helicobacter pylori*), and symptomatic diarrhoeal disease in consumers.

Wastewater treatment markedly reduced the risk of helminth (particularly *Ascaris*) infections to consumers from unrestricted irrigation of vegetables but the level of treatment necessary to reduce risk to an acceptable level could not be determined.

In populations that were exposed to aerosols from sprinkler irrigation, a significantly higher risk of enteric viral and bacterial infections was observed when the wastewater contained at least $10^6$ thermotolerant coliforms ($TCs$) $100$ mL$^{-1}$ but no increased risk was attributed to waters with $10^3$–$10^4$ $TCs$ $100$ mL$^{-1}$ ($TCs$ are used as rough indicators of water contamination with human pathogens).

An increase in symptomatic diarrhoeal disease and enteric viral infections attendant with children in direct contact with wastewater under flood or furrow irrigation (e.g. through labor or play) was apparent for wastewaters with $>10^4$ $TCs/100$ml. The threshold level for symptomatic diarrhoeal disease in adult field workers was $10^6$ $TCs$ mL$^{-1}$.2
Studies on *Ascaris* prevalence suggested that the then current WHO guideline level of ≤ 1 nematode 3 (helminth) eggs L⁻¹ provided inadequate protection of children in communities where children were in close contact with the wastewater, and that a revised guideline of ≤ 0.1 nematode eggs L⁻¹ would be more appropriate in such situations. From the perspective of modern, planned agricultural wastewater irrigation, a pragmatic limitation of the epidemiological approach is that the public, governments and other stakeholders need health risk estimates prior to the commissioning of the project. It is little consolation to find out retrospectively that the scheme was too risky. This can be addressed to some degree by referring to epidemiological studies undertaken elsewhere. But in some cases the transferability of such studies is questionable: marked differences may exist in the pathogen profile of the source water, treatment efficiencies, irrigation methods, crop types, and even consumption behaviour (e.g. the amount of food consumed and the availability of clean potable water to wash food). In contrast, QMRA can be used to develop a risk estimate specific to the wastewater irrigation situation at hand. QMRA is a four-step process involving: (i) hazard identification, (ii) exposure assessment, (iii) dose-response modeling, and (iv) risk characterization (Haas et al., 1999). These steps are described in detail elsewhere (e.g. Haas et al., 1999; Hamilton et al., 2005c; NRMMC et al., 2006). In brief, hazard identification involves determining the pathogens of concern, exposure assessment comprises defining of the exposure pathway so the concentration of the pathogens reaching the consumer can be determined, dose-response modeling defines the probability of infection as a function of pathogen dose, and the final step, risk characterization, brings together the exposure and dose-response models to arrive at an estimate of an adverse outcome, such as infection.

Several QMRAs have been developed for determining risks associated with consumption of vegetables irrigated with wastewater. The earliest such models were deterministic; that is, parameters and inputs were represented by a single value: a point-estimate (Asano and Sakaji, 1990; Asano et al., 1992; Shuval et al., 1997). More recently, the stochastic approach, whereby such point-estimates are replaced with probability distributions, has been preferred as it accounts for uncertainty (Tanaka et al., 1998; van Ginneken and Oron 2000; Petterson et al., 2001; Hamilton et al., 2006a,b). Deterministic modeling has the advantage of simplicity and has therefore been proffered as a pragmatic approach in several major wastewater irrigation guidelines (US EPA and US AID, 2004; NRMMC et al., 2006; WHO 2006). Recently, a computer program (RIRA: Recycled water Irrigation Risk Analysis) was developed that further simplifies the deterministic modeling process for water resource and health officials (Hamilton et al., 2007). RIRA complements the methods outlined in the aforementioned major wastewater irrigation guidelines and hides all mathematical procedures from the user.
Through accounting for uncertainty, stochastic modeling is theoretically superior but is necessarily more complicated, typically requiring Monte Carlo simulation techniques. It has therefore mostly been restricted to the realm of research rather than routine application. In a recent stochastic QMRA for spray irrigation of vegetable crops with non-disinfected secondary treated wastewater, Hamilton et al. (2006a) demonstrated that implementation of a withholding period, wherein conventional water is substituted for wastewater, could be an effective means of mitigating the risk of enteric virus infection to consumers (Table 6). For all combinations of crop type and effluent quality, the estimated annual risks of infection satisfied the commonly-propounded benchmark of ≤10⁻⁴, i.e. one infection or less per 10,000 people per year (USEPA, 1989; Macler & Regli, 1993), providing 14 d had elapsed since irrigation with reclaimed water. It should be noted that when they used a different, less aggressive, viral decay rate constant markedly less protection was conferred, with broccoli and cucumber the only crops satisfying the 10⁻⁴ standard for all effluents after a 14-d withholding period. Stine et al. (2005) also used QMRA to illustrate the value of a 14-d withholding period. In a deterministic model they showed that substantially higher pathogen concentrations could be tolerated in the source water if this withholding period was instigated.

Possible location for Table 6.

To date, most wastewater QMRAs have related to consumption of irrigated produce. There has been only limited effort to model risks arising from other exposure routes, such as inhalation of aerosols (Gardner et al., 1998). QMRA could prove a useful complementary tool to epidemiological studies when trying to quantify occupational and community risks, particularly in developing countries, where workers and children are more likely to be directly exposed to untreated wastewater. Unfortunately, helminth infections are probably the single largest threat to the health of most populations in the developing world (Mara and Cairncross, 1989) but to date no QMRAs have been developed, owing to the absence of a dose-response model (Petterson and Ashbolt, 2003).
Subsequent to the 1999 Stockholm Framework, which promulgated a flexible approach to developing guidelines (Bartram et al., 2001) – i.e. one that accommodates social, cultural, economic and environmental realities –, several major reuse guidelines have been revised and QMRA has been substituted for prescriptive thresholds on pathogen concentrations. For example, QMRA is promoted in a broader risk management context in the World Health Organisation’s new Guidelines for the Safe Use of Wastewater, Excreta and Greywater (WHO, 2006), the new Australian Guidelines for Water Recycling: Managing Health and Environmental Risks (NRMMC et al., 2006), and the American Guidelines for Water Reuse (USEPA and USAID, 2004).

The vadose zone could potentially act as a significant filter protecting aquifers from possible microbial contamination arising from wastewater irrigation. For example, experiments by Lance et al. (1976) suggest that at least 99.99% removal of viruses could be achieved when passing secondary sewage effluent through 250 cm of calcareous sand. Recent research on preferential flow paths, however, suggests that certain soil, wastewater and pathogen combinations could indeed give rise to significant aquifer contamination hazards. For example, a study on transport of *C. parvum* oocysts in variably saturated soil showed that preferential flow via ‘fingers’ was a notable route in disturbed sand columns whereas macropore flow occurred in undisturbed silt loam columns (Darnault et al., 2004). This research also involved construction of a simulation model for oocyst transport by preferential flow, and this model provided reasonable predictions of breakthrough in the sand columns but was ineffective at accounting for breakthrough in the columns with macropores. Stagnitti (1999) constructed a spatially and temporally explicit mathematical model of preferential transport of bacteria through the vadose zone. In contrast to the situation with viruses and protozoan cysts, the possibility of re-growth outside the host had to be accounted for.

Despite these and other advances in understanding pathogen movement through the vadose zone (e.g. Bundt et al. 2001; Assadian et al., 2005; Bai and Lung, 2005; Smith and Hegazy, 2006; Steenhuis et al., 2006), the research is yet to be used in the broader context of risk assessment models for public health protection. Perhaps one of the first steps in this direction was the observation by Darnault et al. (2004) that the number of oocysts transported across their columns (given ‘worst-case’ scenario initial pathogen loads) was ‘several orders of magnitude above an infective dose.’ This is a useful starting point, but in order to calculate risk posed to an individual the QMRA approach described above needs to be applied to contaminated groundwater risks. For example, an exposure model for a scenario where the aquifer is used as drinking water supply would need to account for, among other things, the numbers of pathogens reaching the aquifer, the size of the aquifer and its mixing hydraulics, dilution with other waters prior to reticulation (e.g. river water), the efficiency of any drinking water treatment processes, and drinking water consumption rates. Having derived a pathogen dose in such a manner, an appropriate probabilistic dose-response model for the pathogen (e.g. the exponential for *C. parvum* –Dupont et al., 1995; Haas et al., 1996) could be used to calculate risk, rather than the simpler threshold concept of infective dose. Preferential flow modeling could therefore feed into a broader QMRA model to give us a markedly improved understanding of risks posed by pathogen transport through the vadose zone.
Considering their largely municipal and industrial origin, most wastewaters are not considered to pose a plant pathogen threat to crops. In fact, in most cases it would probably be reasonable to expect conventional surface waters to harbor more pathogens. In contrast, on-site reclamation of wastewater, such as collecting irrigation drainage water or wash-water for further irrigation or washing, does have the potential to concentrate and re-distribute plant pathogens (Hamilton et al., 2005b). Also, if not managed accordingly, the high concentrations of nutrients in most wastewaters could potentially alter plants’ defenses to pathogen attack (see nutrients section above).

Chemicals of Human Health Concern

Pathogenic microorganisms are undoubtedly the prime health concern posed by wastewater irrigation in the developing world. Affluent nations, on the other hand, have witnessed a shift in focus in recent years, with the potential for chronic health effects associated with long-term exposure to chemicals gaining increasing attention. The impressive removal efficiencies of the latest membrane technologies (particularly reverse osmosis), coupled with disinfection treatments, have allayed many, but certainly not all, pathogen-related health fears (Chen et al., 1998; Loge et al., 1998; Wilf, 1998, Nasser et al., 2006). The risks posed by chemicals, on the other hand, are less certain and consequently raise more debate. There are four factors contributing to the uncertainty about chemical risks. First, there is typically a very large number of chemicals to consider, and some toxicants or carcinogens may simply pass undetected. The potential for synergistic and antagonistic effects among chemicals further complicates the picture (Fent et al., 2006). Second, while the removal or breakdown efficiencies of sewage treatment processes are known for many chemicals, there are many for which they are not. Third, chemically-induced health effects are most likely chronic with long latency periods, e.g. cancers, and would therefore be difficult to detect if they were indeed occurring. Finally, transfer from the soil to plant is poorly understood for many chemicals. Overlaying these concerns is the issue of concentration; many of these chemicals are present at very low levels, often below detection limits. The implication is double-edged: concentrations may be so low that the likelihood of negative health effects is negligible or, alternatively, we may be failing to detect potentially hazardous chemicals that could be problematic at extremely low concentrations, given exposure over a long period of time or accumulation in the environment.
Four broad groups can be used to define the chemicals in wastewater of greatest human health concern: heavy metals, pharmaceutically-active compounds (PhACs), endocrine disrupting compounds (EDCs) and disinfection byproducts (DBPs) (Toze, 2006). EDCs are a group of organic and inorganic contaminants that has raised particular concern in recent years. They interfere with the functioning of an animal’s endocrine system, and they can be either naturally occurring or synthetic compounds. The developing human embryo or foetus is probably the most vulnerable stage to the harmful effects of endocrine disruptors. Exposure to much higher concentrations would most likely be necessary to cause harm to adults (e.g. to the reproductive system or body homeostasis) (Moore and Chapman, 2003). A wide variety of EDCs has been found in raw wastewaters and the exact profile clearly depends on the inputs into the sewer. Some of the more common EDCs present in wastewaters include pesticides (e.g. DDT and atrazine), organohalogens (e.g. dioxins and PCBs), alkylphenols, phthalates, hormone drugs (e.g. oestradiol from the contraceptive pill), phenols, aromatic compounds and heavy metals (Toze, 2006; Ying, 2006). Several studies have demonstrated highly effective degradation or removal of specific EDCs by reasonably standard sewage treatment processes (e.g. Staples et al., 1997, 1998; Wang et al., 2003) yet others, including some estrogens, have proved more problematic (de Mes et al., 2006). Moreover, as noted by Ying (2006), a general paucity of monitoring data means that the typical concentrations of many organics in treated wastewater streams are unknown. Another important consideration in the EDC debate is that humans are exposed to many natural compounds with markedly higher endocrine activity than the manufactured chemicals discharged to sewers (Mazur and Adlercreutz, 1998; Moore and Chapman, 2003; Cunliffe, 2006).
Consumer Perceptions

In developing countries wastewater irrigation is rarely a matter of choice: livelihoods are dependent upon access to irrigation water, and wastewater is often the only available source. In wealthy countries, on the other hand, the public expresses its voice and influence on food production systems and water resource management. In such situations there is little point in understanding biophysical issues of wastewater irrigation if the practice is not supported by the public. Community perception on wastewater reuse has been the subject of recent reviews by Hartley (2003), Po and Nancarrow (2004) and Po et al., (2004). A common theme emerging from these reviews is that the level of acceptance generally decreases as the degree of contact or proximity to the reclaimed water increases. The implication is that wastewater irrigation of food crops tends to be preferred over potable reuse but is considered less acceptable than more distant uses such as landscape irrigation or heavy industry. For example, in collating data from several American and Australian studies Po et al. (2004) found that opposition to reuse ranged from 44–77% for drinking water, 7–21% for vegetable crop irrigation (USA studies only), and 2–5% for golf course irrigation.

Opinion polls give only a superficial understanding of the public’s position on wastewater reuse. They provide negligible insight into the reasons for acceptance or rejection – necessary knowledge for appropriate long-term planning (Syme and Nancarrow, 2006). Accordingly, recent progress in the social psychology of wastewater reuse has seen the application of attitudinal modeling techniques to particular reuse schemes. Po et al. (2005) constructed a model around Ajzen’s (1985) theory of planned behavior and applied it to the then proposed vegetable wastewater irrigation scheme at Werribee, on the outskirts of Melbourne, Australia. This involved interviewing a socio-economically stratified cross-section of Melbournians and running the data through the model. Attitudes, subjective norms (the effects of others on one’s intentions), trust, and emotions (particularly disgust) emerged as the key explanatory variables for predicting behavior. Perceived control, risk perceptions and feelings of environmental obligation were important lesser predictors. The relatively modest predictive power of risk perception was an unexpected finding. Another noteworthy finding was that knowledge of the reuse scheme did not emerge as a statistically significant predictor in the model. This somewhat counterintuitive finding has significant implications for the management of the scheme: heavy investment in communication and education programs may be largely futile. It is interesting to note that when the model was applied to an indirect potable reuse scenario (aquifer recharge) in Perth, Western Australia, perceived risk once again failed to emerge as a dominant predictive variable, and the contribution of knowledge was again statistically insignificant.
Most of the detailed research on consumer perceptions of wastewater irrigation relates to just a handful of countries, most notably Australia and the USA (Hartley 2003; Po and Nancarrow 2004; Po et al., 2004). Understanding consumer attitudes in different cultures will become increasingly important, particularly in countries that are increasing in affluence. For many of the world’s poor, however, wastewater irrigation will remain an issue of no choice.

CONCLUSION

Wastewater irrigation is a widespread practice, particularly in parts of the world where freshwater is limiting, most notably the Near East, but also in relatively wet regions, where environmental effects associated with wastewater discharge to oceans and rivers are an incentive for its reclamation. Much of the wastewater irrigation takes place in the peri-urban fringe of large cities, where there is ready access to municipal wastewater and produce markets. Continuity of supply makes wastewater irrigation an attractive proposition, and in many situations in the developing world it is indeed the only water available. Unfortunately, this has led to the widespread use of poorly-treated, or even raw, wastewater, which in turn has had significant public health consequences; but it must be remembered that in many cases livelihoods are highly dependent upon wastewater irrigation: it is not a matter of choice.
The following remain as particularly significant gaps in the science of sustainable wastewater irrigation:

- long-term accumulation of bio-available forms of heavy metals in soils; 3
- an understanding of the balance of various factors affecting the environmental fate of organics in wastewater irrigated soils; 4
- the influence of reuse schemes on catchment hydrology, including transport of salt loads; 6
- risk models for helminth infections (mostly pertinent to developing nations); 7
- application of public health microbial risk assessment models to wastewater contamination of aquifers and surface-waters used by humans; 9
- transfer efficiencies of chemical contaminants to plants; 10
- effects of chronic exposure to chemical contaminants through consuming wastewater-irrigated food; 11
- detailed understanding of the psychology and sociology of wastewater irrigation, particularly in different cultures. 13

While there are many other issues relating to sustainable wastewater irrigation, such as on-site management of salts and nutrients, reasonably sufficient knowledge of these exists and they can usually be effectively managed. While beyond the general scope of this review, we must also recognize that the challenges facing sustainable wastewater irrigation extend well beyond science. Not only do some countries lack sufficient sewerage infrastructure, but for many institutional and water policy reform will be necessary before sustainable wastewater irrigation becomes a reality.

ACKNOWLEDGEMENT

This research was funded by the Australian Research Council (Linkage Project 0455383). We thank Yvonne Rogers for formatting the manuscript.

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United States Salinity Laboratory. 1954. Diagnosis and improvement of saline and sodic soils. USDA Handbook No. 60. US Government Printing Office, Washington, DC, USA.


FIGURE CAPTIONS

**Figure 1.** Heavy metal concentrations in Western Treatment Plant soils as a function of the length of time of sewage irrigation. Note that ordinate scale needs to be divided by 100 to obtain Cd concentrations in mg/kg. Reprinted with permission from Xiong et al. (2001).

**Figure 2.** Salt, soil structure and solute transport interactions in relation to crop productivity and potential environmental impacts. Developed from the model of So and Aylmore (1993) and reprinted with permission from Maher et al. (2004).
Table 1. Sanitation, sewerage, and wastewater treatment statistics for major world regions. LA&C = Latin America* and Caribbean. Data from WHO and UNICEF (2000).

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*WHO and UNICEF (2000) included non-Latin South American countries, Guyana and Suriname, in the LA&C category. Likewise, their Northern America category is not synonymous with North America, as it excludes Caribbean countries and Mexico.
Table 2. Wastewater irrigation statistics for selected countries (or provinces/states or cities). Where the estimate is known to be based on data collected at least 4 years earlier than the publication date of the citation, the year pertaining to the estimate is given in italics after the country name. Some estimates may include minor contributions from sectors other than irrigation. See source references for specifics, qualifications and limitations relating to estimates. Some estimates have been derived from other statistics presented in the source publications.
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Table 3. Percentage use of reclaimed wastewater (by volume) for different purposes. After RCC and WCIWRWG (2003).

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| Country (region/city) | Land area irrigated | % of all agriculture | Vol. of wastewater | % of et reclaimed |
Table 4. General classification scheme for saline and sodic soils. ESP = Exchangeable sodium percentage, ECe = Electrical conductivity in saturation paste extract. From United States Salinity Laboratory (1954).

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Table 5. Metal bioavailability grouping. From Hamilton et al. (2005a) and based on Chaney (1980).

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Table 6. The annual probability of enteric virus infection associated with consuming vegetables that have been spray-irrigated with secondary effluent from four different treatment plants – Effluent A = MRWPCA, B = OCSD AS, C = OCSD TF, D = Pomona AS (see Tanaka et al., 1998). Savoy, Grand Slam and Winter Head are cabbage cultivars. ‘Time’ is the length of the wastewater withholding period in days. Each value is the upper 95% confidence limit of the mean (UCL0.95). See Hamilton et al. (2006a) for details of the model.

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UNSTABLE SOIL