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Farm-Based Measures for Reducing Human and Environmental Health Risks from Chemical Constituents in Wastewater

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Abstract

There is a significant imbalance between the number of publications describing potential and actual environmental and health impacts from chemically contaminated wastewater, and reports outlining concrete options to minimize the related risks where conventional wastewater treatment is not available. This gap applies more to inorganic and organic contaminants than excess salts or nutrients. This chapter outlines some of the options available that could be considered in and around the farm, looking at heavy metals, salts, excess nutrients and organic contaminants. The emphasis is placed on low-cost options applicable in developing countries. While such measures can reduce negative impacts to a certain extent, it remains crucial to ensure that hazardous chemicals are replaced in production processes; industrial wastewater is treated at source and/or separated from other wastewater streams used for irrigation purposes; and fertilizer application rates and related possible subsidies adjusted to avoid over-fertilization.

INTRODUCTION

Where irrigation with untreated, partly treated or diluted wastewater cannot be avoided or is otherwise common, negative impacts on irrigated crops, soils and groundwater that can affect human and environmental health are likely (Ayers and Westcot, 1985; Murtaza et al., 2009; Pescod, 1992; Pettygrove and Asano, 1985; WHO, 2006b). Several chapters in this book focus on pathogenic threats, related risk assessments and risk mitigation. This chapter has its focus on non-pathogenic contaminants. As outlined in Chapter 6, aside from organic chemicals, debris and solutes, non-pathogenic components of polluted irrigation water can comprise a range of elements that can be essential plant nutrients, undesirable salts or metals and metalloids in toxic concentrations, depending on their concentration and solubility.

The high concentrations of chemical constituents that need to be addressed in wastewater-irrigated environments can be roughly divided into:

- metals and metalloids, such as cadmium (Cd), chromium (Cr), cobalt (Co), molybdenum (Mo), nickel (Ni), zinc (Zn), lead (Pb), arsenic (As), selenium (Se), mercury (Hg), copper (Cu) and manganese (Mn), among others;
- nutrients such as nitrogen (N), phosphorous (P), potassium (K), calcium (C) and magnesium (Mg), which in high concentrations might suppress other nutrients and/or affect plant growth and aquatic life;
- salts and specific ionic species such as sodium (Na), boron (B) and chloride (Cl);
- persistent organic pollutants (POPs), such as pesticides as well as so-called emerging contaminants, like residual pharmaceuticals, endocrine disruptor compounds and active residues of personal care products.

To avoid potential negative impacts, conventional wastewater-treatment options, which can control the release of most of these contaminants into the environment, remain the key to protecting water quality for beneficial uses including agriculture.

In theory, it could be expected that, with increasing economic development and industrialization, treatment standards, regulations and capacities grow concomitantly, allowing a society at each development stage to deal with its own waste. However, there are many development pathways, and growth in each sector of the economy does not always run in parallel. The so-called emerging economies or markets are a good example of this process. China, India, Pakistan and Mexico are some of the largest countries in this group, but they are also those most often cited for large-scale industrial water pollution and irrigation with highly polluted water (Jiménez and Asano, 2008). Many other low-income countries show, at a smaller scale, similar challenges of emerging industrial sectors or mining activities while institutional, technical and/or regulatory capacities for wastewater treatment are not yet in place. The result is a situation in which not only microbiological contaminants, but also industrial effluent, pose a threat to farmers and consumers of wastewater-irrigated food. The related possible environmental and health impacts are described in a range of papers (Abaidoo et al., 2009; Hamilton et al., 2007; Stevens and McLaughlin, 2006) but they are usually brief in answering what could be done where appropriate conventional treatment facilities are missing. This chapter tries to address the gap by outlining some options for non-pathogenic contaminants including salts.

METALS AND METALLOIDS

All of the potentially toxic metals are naturally present in the environment in trace amounts and are ingested with food, water and air. Human bodies have the ability to deal with these background levels. The World Health Organization (WHO) has established guidelines on allowable consumption of various toxins (WHO, 2006a) and guidance values in irrigation water (WHO, 2006b). Several of these metals and metalloids are of particular concern due to their adverse effects on agricultural productivity as well as environmental and human health. In a review of wastewater use in the Australian horticultural production industry, Hamilton et al. (2005) classified potentially phytotoxic metals in wastewater into four groups based on their retention in soil, translocation in plants, phytotoxicity and potential risk to the food chain (Table 11.1). They categorized Cd, Co, Se and Mo as posing the greatest risk to human and animal health because they may accumulate in crops without damaging them. Indeed, the visible symptoms of toxicity vary from plant to plant, even if they contain elevated concentrations of toxic metals and metalloids (Clemens, 2001). The recent the guidelines of the WHO also consider Cd to be of particular concern because of both high levels of toxicity and bioaccumulation in crops (WHO, 2006b).

Metals such as Cd, Hg and Pb do not have any essential function but they are detrimental, even in small quantities, to plants, animals and humans, and accumulate because of their long biological half-life (Goethberg et al., 2002). Other metals and metalloids, such as Mn, Zn, B and Cu are essential micronutrients in small concentrations, but harmful to crops in higher concentrations. Some, such as Cu and Zn, become toxic to plants before they reach high enough concentrations to be toxic to humans, thus plants function here as a barrier mitigating potential health risks (Hamilton et al., 2005; Johnson, 2006).

Although wastewater treatment is the best choice in managing wastewater in agriculture, the costs involved in engineering-based technologies for wastewater treatment are prohibitively high for most developing countries. Even where wastewater treatment plants are externally funded, they usually only treat a small fraction of the wastewater produced and, depending on their type, can face significant maintenance problems. However, some farm-based measures and low-

Table 11.1 *Metal bio-availability grouping*

Group	Metal	Soil adsorption	Phytotoxicity	Food chain risk
1	Ag, Cr, Sn, Ti, Y and Zr	Low solubility and strong retention in soil	Low	Little risk because they are not taken up to any extent by plants
2	As, Hg and Pb	Strongly sorbed by soil colloids	Plant roots may adsorb them but not translocate to shoots; generally not phytotoxic except at very high concentrations	Pose minimal risks to the human food chain
3	B, Cu, Mn, Mo, Ni and Zn	Less strongly sorbed by soil than Groups 1 & 2	Readily taken up by plants and phytotoxic at concentrations that pose little risk to human health	Conceptually the 'soil–plant barrier' protects the food chain from these elements
4	Cd, Co, Mo and Se	Least of all metals	Pose human and/or animal health risks at plant tissue concentrations that are not generally phytotoxic	Bioaccumulation through the soil–plant–animal food chain

Source: From Hamilton et al. (2005)

cost treatment options can reduce the risk to the environment and human health (WHO, 2006b).

The key steps to follow are:

- identifying which geographical areas have elevated risk based on consideration of potential metal sources;
- quality-assured testing of soil and plant samples to verify the level of risk;
- identifying alternative varieties of the same desired crop that take up the least metal or convert the toxin to less toxic forms when grown in high-risk areas;
- developing irrigation, fertilization and residue management strategies that help to minimize metal uptake by plants;
- recommending cultivation of other crops with lower health risk (crop restrictions) if the measures mentioned above fail to safeguard humans;
- zoning affected areas for non-agricultural land use or land rehabilitation.

Most knowledge refers to the last option and industrially contaminated sites in developed countries where the affected land has a high value and costs of remediation are met by the state or by the polluter. In these cases, in situ and ex situ engineering options are applied (Table 11.2).

However, within the economic constraints of developing countries and in terms of farm-based strategies aimed at addressing wastewater-induced contamination of metal/metalloids, viable risk-reduction options can be categorized as:

Table 11.2 *In situ and ex situ engineering options adopted for remediated metal/metalloid contaminated soils*

Element	Method/Treatment/Amendment	References
Cd, Zn, As, Ti, Pb, Cu, Cr	Removal and replacement of contaminated soil	Iimura (1981)
	Containment: caps, vertical barriers, etc.	USEPA (1997)
	Solidification/stabilization: cement-based, polymer-microencapsulation, vitrification	Dutré et al. (1998); USEPA (1997)
	Separation/concentration: soil-washing, soil-flushing	USEPA (1997)
	Electrokinetics	Virkutyte et al. (2002)
Cd, Mn, Ti, Cr	Microwave immobilization	Abramovitch et al. (2003)
Cd, Cu, Pb, Zn	Suphidization pre-treatment and Denver floatation	Vanthuyne and Maes (2002)

- Soil-based treatment with non-toxic amendments to form insoluble complexes of metals and metalloids, rendering their availability at low concentrations in the root zone.
- Plant-based strategies for soils and waters contaminated with metals and metalloids through the cultivation of specific plant species capable of accumulating target ionic species in their shoots, thereby removing them from the soil or water. These mechanisms include phytoremediation (including hyperaccumulation and phytomining), chelate-enhanced phytoextraction and the use of transgenic crops.

Soil-based treatment

Hamilton et al. (2007) describe increasing total heavy metal concentrations in soils irrigated with sewage for up to a century. The authors also found that potentially bio-available forms of the metals have increased. However, the authors also report that plant tissue showed relatively low concentrations as the metals were strongly absorbed in the soil. Steering the processes that limit the solubility and plant availability of heavy metals and metalloids in soils is possible, e.g. through the use of soil amendments including gypsum, lime ($CaCO_3$), phosphate materials, hydrous Fe and Mn oxides, clay minerals and organic matter (Table 11.3).

These amendments have been shown to immobilize metals and metalloids through:

- formation of insoluble metal phosphate minerals;
- sorption of contaminants on Fe and Mn oxide surface-exchange sites, co-precipitation – formation of contaminant Fe and Mn compounds;
- sorption of contaminants on exchange sites of organic materials including manures, composts and sludges;

Table 11.3 *Soil amendments utilized for the in situ immobilization of metals and metalloids*

Element	Method/Treatment/Amendment	References
Pb	Hydroxyapatite (HA)	Chlopecka and Adriano (1997); Zhu et al. (2004)
Cd	Alkaline biosolids, lime-stabilized biosolids	Basta et al. (2001); Wong et al. (2004)
Cd/Zn	Sepiolite	Alvarez-Ayuso and García-Sánchez (2003)
Ti, Zn, Cd, Mn, Pb, Hg and Co	Zeolite (natural and synthetic)	Chlopecka and Adriano (1997); García-Sánchez et al. (1999); Haidouti (1997); Malliou et al. (1994); Oste et al. (2002)
Pb	Phosphoric acid (H_3PO_4) and calcium dihydrogen phosphate ($Ca(H_2PO_4)_2$)	Brown et al. (2004); Chen et al. (2003); Melamed et al. (2003)
Cd and Pb	Iron oxide waste by-product	Chlopecka and Adriano (1997)
Cd, Pb and Zn	Di-ammonium phosphate (DAP)	McGowen et al. (2001)
Pb	Phosphate rock	Basta et al. (2001); Hettiarachchi et al. (2001)
Pb, Cd, Zn	Triple super phosphate (TSP)	Hettiarachchi et al. (2001); Hettiarachchi and Pierzynski (2002)
Cd, Pb and Zn	Phosphate clay	Singh et al. (2001)
Pb	Mn oxide	Hettiarachchi and Pierzynski (2002)
Cd	Liming	McLaughlin and Singh (1999)
Cr (Cr(VI) reduction to Cr(III))	Organic amendments	Bolan et al. (2003)
Ni	Limestone	Kukier and Chaney (2001)
As	Simultaneous addition of lime and $FeSO_4$	Warren et al. (2003); Warren and Alloway (2003)
As	Goethite	Garcia-Sànchez et al. (1999)
As	Water treatment sludges and red mud	Lombi et al. (2004)

- sorption of contaminants on mineral surface-exchange sites or incorporation into the mineral structure of zeolites, natural aluminosilicates and aluminosilicate by-products.

The aforementioned amendments form insoluble complexes of metals and metalloids, reducing their availability at low concentrations in the root zone and reducing their assimilation by plants (Hussain, 2000; Zhu and Alva, 1993).

Although soil-based management via addition of amendments to immobilize metals/metalloids offers great opportunity to minimize element bio-availability, practical limitations must be considered. These include the management of sites co-contaminated with several elements; cost and availability of amendments; cost of long-term monitoring programmes; and suitability to particular soil and climatic conditions. Care should also be taken in the post-management phase, particularly if the site is exposed to acidic water (low pH) which may transform insoluble complexes into soluble forms.

Plant-based treatments

Soils contaminated with metals and metalloids can be improved through the use of certain plant species. This approach is broadly known as phytoremediation (Chaney et al., 2007; Cunningham et al., 1995; Salt et al., 1996). As an important category of phytoremediation, phytoextraction involves the use of pollutant-scavenging plants to absorb and concentrate metals and metalloids from the soil into above-ground biomass, which may be harvested to remove the elements from the field (Table 11.4) . Plants able to accumulate high concentrations of metals are known as hyperaccumulators (Box 11.1).

BOX 11.1 HYPERACCUMULATORS

Three internationally recognized hyperaccumulator definitions are used to describe the efficiency of phytoextraction for a given metal or metalloid, namely:

- Translocation Factor;
- Extraction Coefficient;
- Bioaccumulation Factor.

The Translocation Factor or shoot/root quotient is defined as the ratio of a given heavy metal in plant shoots as compared with that in the plant root. A Translocation Factor >1.0 indicates preferential partitioning of metals to the shoot (Baker and Whiting, 2002; Branquinho et al., 2007; González and González-Chávez, 2006). The Extraction Coefficient has been described as the heavy metal concentration in the shoot divided by the (total) heavy metal concentration in soil and can be used to evaluate the ability of a plant to accumulate a heavy metal (Branquinho et al., 2007; Chen et al., 2004). Finally, the Bioaccumulation Factor is defined as the ratio of metal concentration in plant shoots to the extractable concentration of metal in the soil and is used for the quantitative expression of accumulation (Branquinho et al., 2007; Derem et al., 2006).

The concentrations of metals accumulated in hyperaccumulator plants may be 100 times greater than those occurring in non-accumulator plants growing on the same substrates (Chaney et al., 2007). Currently, there are more than 400 plant species categorized as hyperaccumulators of metals and metalloids (Cobbett, 2003).

Table 11.4 *Selected case studies on phytoremediation*

Element	Plant Species	Reference
As	*Pteris vittata* L. and *Pityrogramma calomelanos*	Francesconi et al. (2002); Tu and Ma (2002); Wongkongkatep et al. (2003); Zhang et al. (2002)
Cd/Zn	*Thlaspi caerulescens*	Brown et al. (1994, 1995a, 1995b); Lombi et al. (2001); Schwartz et al. (2003)
Ni	*Alyssum murale*, *Phyllanthus serpentinus*, *Berkheya coddii*	Abou-Shanab et al. (2003); Chaney et al. (2007); Kersten et al. (1979); Robinson et al. (1999)
Se	*Astragalus racemosus*	Parker et al. (1991)
Mn	*Alyxia rubricaulis*, *Phytolacca acinosa* Roxb.	Brooks et al. (1981); Xue et al. (2004)
Ti	*Biscutella laevigata*, *Iberis intermedia*	Anderson et al. (1999)
Cu	*Aelanthus biformifolius*, *Haumaniastrum katangense*	Brooks (1977); Brooks et al. (1978)
Co	*Haumaniastrum robertii*	Brooks et al. (1978)

Because the costs of growing a phytoremediation crop are minimal as compared to those of soil removal and replacement, the use of plants to remediate hazardous soils is seen as having great promise (Chaney et al., 2007). This is particularly pertinent for elements that may provide economic phytomining potential (Ni, Co, Ti and Au). Following harvest of the metal-enriched plants, their weight and volume can be reduced by burning the dried biomass which results in a high-grade ‘metal ore’.

Chelate-enhanced phytoextraction utilizing ethylenediaminetetraacetic acid (EDTA) and high biomass producing plant species such as *Brassica juncea* (L.) Czern (Indian mustard) has also been investigated (Kumar et al., 1995). However, an observed drawback was the equally enhanced leaching of Pb down the soil profile (Greman et al., 2003; Madrid et al., 2003; Römkens et al., 2001; Wu et al., 2004).

In addition to phytoextraction, phytoremediation can also be achieved through reduction in the bio-availability of metals in the soil (phytostabilization), volatilization of pollutants such as Hg and Se from the foliage (phytovolatilization) and removal of contaminants by plant roots from flowing water (rhizofiltration) (Pilon-Smits, 2005). Rhizofiltration is particularly effective in applications where low metal concentrations and large volumes of water are involved (Salt et al., 1996).

However, phytoremediation has certain limitations which need to be addressed in general and on a site- and contaminant-specific basis. These include:

- Phytoextraction of metals and metalloids may take years/decades which limits its practical applicability.
- It is restricted to sites where the concentration of the contaminants (or co-contaminants) are not toxic to the plants proposed for phytoremediation.
- A specific phytoremediation 'prescription' cannot be applied to every site with a certain chemical contaminant because different site-specific conditions may not be suitable for the target plant.
- In situ phytoremediation is often restricted to sites conducive to growth of the selected plant with the contaminant located within the root zone.
- It is limited by bio-availability of pollutants, only a fraction of which may be bio-available but regulatory clean-up standards require that all the pollutant is removed. In this scenario phytoremediation may not be applicable.

Crop choice and crop restriction

As described above, crops vary in their absorption behaviour and thus risk potential for humans. In addition, some crops are consumed in larger quantities than others and some are only used as fodder plants and might not enter the human food chain. Thus, crop selection can contribute to decreasing human health risks. For example, in the case of irrigation with untreated wastewater, leafy vegetables accumulate certain metals such as Cd in greater amounts than non-leafy species (Qadir et al., 2000). Bellows (1999) gives as a rule of thumb a heavy metal absorption ratio of 1:10 for fruits and seeds versus leaves and roots. This favours cereals, legumes like beans and peas, tomatoes or fruits over vegetables such as lettuce, cauliflower, carrots or spinach. However, consideration must be given to the quantities of e. g. rice or leafy vegetables actually consumed, and hence contribution to dietary intake of the metal or metalloid, before farmers are challenged to change their cropping pattern. There is a strong relationship between the long term consumption of Cd-contaminated rice and human Cd disease (Kobayashi et al., 2002; Nordberg, 2003).

A shift in crop choice is only feasible and sustainable if there is a market and comparative market value for the alternative crop, unless subsidies are provided. Changed cropping practices might also require additional training and different tools, or even long-term tenure security if, for example, tree crops are recommended. Crop restrictions can therefore be hard to implement if necessary conditions are not in place. There are, however, examples of successful or partly successful implementation of crop restriction in wastewater use schemes in several countries such as India, Mexico, Peru, Chile, Jordan and Syria (Blumenthal et al., 2000; Qadir et al., 2007b). However, the probability of success appears much lower in sub-Saharan Africa and other countries where wastewater irrigation is not confined to (regulated) irrigation schemes but takes place along polluted streams and thus remains informal.

Zoning

Where there are no further options to maintain the farm, the affected areas might have to be mapped and taken out of production. Simmons et al. (2009) developed a General Linear Regression Model to predict the spatial distribution of soil Cd in a Cd/Zn co-contaminated cascading irrigated rice-based system in Thailand. Preliminary validation indicated that the model can predict soil Cd based on minimal soil sampling and the field's proximity to primary outlets from in-field irrigation channels and subsequent inter-field irrigation flows. Previous research (Simmons et al., 2005) and subsequent health studies confirming Cd-induced renal dysfunction in the exposed population (Swaddiwudhipong et al., 2007; Teeyakasem et al., 2007) also demonstrated the validity of assessing health risks through monitoring Cd intake via dietary exposed pathways in comparison to the Joint FAO/WHO Expert Committee on Food Additives (JECFA) Provisional Tolerable Weekly Intake values established for Cd. While Cd is of high risk, as stated above, soil sampling alone might not be a sufficient indicator of the actual health risk. This is reiterated in the example of arsenic (Box 11.2). However, zoning and taking contaminated areas out of food production should be accompanied by adequate compensation for farmers /landowners or alternative income-generating livelihood opportunities, associated with training and assured markets or subsidies.

BOX 11.2 THE CASE OF ARSENIC

Sources of arsenic contamination in rice fields include geologic soil materials that are naturally high in arsenic; irrigation with contaminated groundwater; residual arsenical pesticides; or application of poultry manure from chickens treated with arsenical antiparasite food additives. In Bangladesh, which has widespread geologic arsenic contamination, the many documented cases of arsenic poisoning have been caused by consumption of contaminated drinking water, not food, although arsenic is of more concern in rice than in other grain crops because flooded soil conditions make arsenate, which mimics the plant nutrient phosphate, more available to plants. However, far more arsenic accumulates in leaves than in grain and, according to Johnson (2006), experiments have so far failed to measure arsenic concentrations above published safe limits in rice grain, even in very contaminated soil. This situation may have changed. Williams et al. (2006) predicted that a daily consumption of rice in Bangladesh with a common total arsenic level of 0.08µg As g^{-1} is similar to a drinking-water intake with the allowed arsenic concentration of 10µg per litre. Meharg et al. (2008) reported that inorganic arsenic is in particular elevated in the bran layer of unpolished (brown) rice and less in white rice. According to FAO, planting rice in raised beds around 15cm above the ground and not in conventional flooded fields counteracted yield losses and resulted in lower arsenic levels in crops and in the soil, as a pilot field study in Bangladesh revealed (Duxbury et al., 2007).

NUTRIENTS IN EXCESS

Wastewater usually contains valuable plant nutrients, such as N, P and K. Depending on whether raw or diluted wastewater is used, the concentrations of the nutrients can vary significantly and might reach levels that can replace fertilizers or are in excess of crop needs and, if biased to certain nutrients, might affect others. Although availability of these nutrients is considered to be a driving force for wastewater irrigation in some developing countries, managing appropriate levels of nutrients in wastewater is a challenging task. Related studies usually encounter a variety of challenges which reduce the management options for farmers.

In general, nutrients in irrigation water are immediately available to the crop, as long as they remain dissolved in the water and soil solution, but may be rendered less available by several soil processes. Some processes result in permanent loss (leaching, volatilization and erosion) and others in nutrient accumulation in the soil (microbiological immobilization, adsorption and precipitation). Hence the proportions of nutrients taken up by plants are different from the proportions of nutrients applied via wastewater (or fertilizers). Because soils and wastewater seldom contain nutrients in optimum ratios, guidelines are needed to optimize wastewater irrigation. A related concept has been presented by Janssen et al. (2005). It requires, however, information on nutrient levels in water, soils and plants, which may not be readily available to resource-poor wastewater farmers or relevant government departments unless obtained through site-specific field trials.

To avoid excessive or unbalanced additions of particular nutrients to wastewater-irrigated soils and crops, farmers can select crops which are less sensitive to high nutrient levels or which can take advantage of high amounts of P and N. Higher N-levels are thus more welcome in farms specializing in leafy vegetables than grains. In addition, fodder grass is well suited to wastewater-irrigation and acts as a scavenger for N and P applied via wastewater. Reduction efficiencies of 84 per cent for N and 54 per cent for P have been reported from wastewater irrigated pastures in Zimbabwe (Nhapi et al., 2002). However, land- and soil-based options depend not only on the type of crop but also local soil and site conditions. Medium- to fine-textured soils, for example, may hold more nutrients than sandy soils, thereby releasing lower quantities in the water percolating through the soil and adding to the groundwater. Groundwater-quality monitoring is required where groundwater is shallow and used for drinking purposes.

Where farmers do not have the option to grow crops which benefit from high N or P levels, the irrigation water might first pass through other systems to transform part of its nutrient load into biomass. This could be an on-farm pond covered with duckweed or a wetland system, like the traditional tank cascades found in Sri Lanka (Awuah et al., 2004; Mahatantila et al., 2008; Nhapi, 2004). In all of these cases, however, it is necessary to remove the net biomass growth in order to prevent eventual decay of the biomass and re-release of the nutrients (Strom, 2006).

Observations from larger urban settings in developed countries show that effluent treatment by land application for cropping and forestry is often less economical than other treatment techniques. This might be due to the increasing economic land value near cities, but in particular the need in temperate climates to cater for the cold season when soils might be sealed by ice, with plants not growing or in dormant state (Jayawardane et al., 2001). In addition, where soils have restricted internal drainage capacity, soil degradation can occur through waterlogging and salinization (Jayawardane et al., 2001; Su et al., 2005). Hence most land-disposal processes are dependent on freely draining soils and the existence of some diversion structure to store effluent during periods of low absorption capacity or plant water demand.

To overcome the constraints associated with conventional land disposal of wastewater in Australia, the Filtration and Irrigated Cropping for Land Treatment and Effluent Reuse (FILTER) technique was developed for the treatment and reuse of secondary sewage effluent (Gardner et al., 2001; Jayawardane, 1995). The FILTER technique combines the use of nutrient-rich wastewater for intensive cropping with biological and physio-chemical filtration through the soil to a subsurface drainage system. It was initially tested on eight 1-ha experimental plots and subsequently trialled on four (4-ha) commercial-scale plots. FILTER plots were constructed by deep ripping to around 1m depth and installing the subsurface drainage system at this depth. The sewage effluent was applied as flood irrigation at the top end of the FILTER plots. Besides nutrient removal, other beneficial effects were reduced suspended solids, oil and grease, and an increased N/P ratio in the drainage water (Blackwell and Arakel, 2004). An obvious disadvantage is the cost factor and equipment required for the set-up of the system, even at smaller scale. However, there might be options for low-cost adaptations.

In cases where there are excess nutrient levels such as N or salts (see below), wastewater can be diluted with freshwater, where possible, to decrease the nutrient concentration and increase the benefits through a higher volume of irrigation water. This option might have a strong seasonal dimension and is only possible where wastewater streams are separated from other surface-water bodies. Where freshwater is not available, the quantity of wastewater applied per unit area can be decreased. The same applies to wastewater with high levels of organic matter. In this case, wastewater should not be applied continuously to allow soil to biodegrade organic matter.

SALTS AND SPECIFIC IONIC SPECIES

Wastewater contains more soluble salts than freshwater because salts are added to it from different sources (Qadir et al., 2007b). There are no economically viable means to remove the salts once they enter wastewater because the techniques, such as cation exchange resins or reverse-osmosis membranes, are prohibitively expensive and are only used to produce high-quality recycled water (Toze, 2006a).

For remediation purposes, wastewater can be divided into: saline wastewater containing excess levels of soluble salts; sodic wastewater characterized by excess levels of sodium (Na^+); and saline-sodic wastewater having both salts and Na^+ in excess concentrations.

The last category is most prevalent. Salinity in wastewater is characterized by its electrical conductivity (EC) expressed in terms of deci-Siemens per metre ($dS\ m^{-1}$). Sodicity is assessed by sodium adsorption ratio (SAR), which is expressed as the relative amounts of Na^+ to that of divalent cations, calcium (Ca^{2+}) and magnesium (Mg^{2+}).

For long-term irrigation with saline and/or sodic wastewater, there is a need for site-specific preventive measures and management strategies, which may include:

- appropriate selection of crop or crop variety capable of producing profitable yield with saline wastewater;
- selection of irrigation methods to reduce crop exposure;
- application of wastewater in excess of crop water requirement (evapotranspiration) to leach excess salts from the root zone;
- wastewater irrigation in conjunction with freshwater, if available, through cyclic applications and/or blending;
- in the case of salt-sensitive crops, via careful seedbed preparation and planting techniques;
- in the case of highly sodic wastewater, through the application of Ca^{2+} (e.g. via gypsum or alternative calcium-rich wastewater) to mitigate Na^+ effects on soils and crops.

Crop selection and diversification

Research efforts have led to the identification of a number of field crops, forage grasses and shrubs, biofuel crops, fruit trees and agroforestry systems which can suit a variety of salt-affected environments and local or regional markets (Maas and Grattan, 1999; Qadir et al., 2008). Salt tolerance depends on several soil, crop and climatic factors and is generally divided into four classes: sensitive; moderately sensitive; moderately tolerant; and tolerant. Relative salt tolerance threshold values for a range of crops as a function of average root-zone salinity are given in Table 11.5. Absolute tolerances will, however, vary depending on climate, soil conditions and cultural practices.

The genetic diversity among these crops provides a range of cropping options, especially as salinity tolerance often varies between different varieties of the same crop. For some crops particular salt-tolerant varieties have been created. Local extension officers and crop-research institutes will be able to provide advice on their in- and output markets.

Table 11.5 *Yield potentials of some grain, forage, vegetable and fibre crops as a function of average root-zone salinity*

Common name	Botanical name	Yield potential (%) at specified salinity (dS m^{-1})		
		50%	80%	100%
Durum wheat	*Triticum durum* Desf.	19	11	6
Barley	*Hordeum vulgare* L.	18	12	8
Cotton	*Gossypium hirsutum* L.	17	12	8
Rye	*Secale cereale* L.	16	13	11
Sugar beet	*Beta vulgaris* L.	16	10	7
Wheat	*Triticum aestivum* L.	13	9	6
Purslane	*Portulaca oleracea* L.	11	8	6
Sorghum	*Sorghum bicolor* (L.) Moench	10	8	7
Alfalfa	*Medicago sativa* L.	9	5	2
Spinach	*Spinacia oleracea* L.	9	5	2
Broccoli	*Brassica oleracea* L.	8	5	3
Egg plant	*Solanum melongena* L.	8	4	1
Rice	*Oryza sativa* L.	7	5	3
Potato	*Solanum tuberosum* L.	7	4	2
Maize	*Zea mays* L.	6	3	2
Carrot	*Daucus carota* L.	6	3	1

Source: Based on the salt-tolerance data of different crops and percentage decrease in yield per unit increase in root-zone salinity in terms of dS m^{-1} as reported by Maas and Grattan (1999)

Crop-diversification systems based on salt-tolerant plant species are likely to be the key to future agricultural and economic growth in regions where saline wastewater is used for irrigation. Such systems, linked to secure markets, should support farmers in finding the most suitable and sustainable crop-diversifying systems to mitigate any perceived production risks, while ideally also enhancing the productivity per unit of saline wastewater and protecting the environment. In all cases, farmers are encouraged to test the actual performance of suggested varieties on their fields.

Irrigation method

There are different ways to irrigate crops, such as surface or flood irrigation, manual irrigation with watering cans, furrow irrigation, sprinkler irrigation and micro-irrigation such as drip or trickle irrigation. Some are more suitable for saline water or other types of low-quality water than others. The clogging of drip irrigation systems is an example. Another one is sprinkler irrigation which may cause injury to crops from the sodium and chloride salts absorbed directly through wetted leaf surfaces, especially where climatic conditions favour evaporation (Ayers and Westcot, 1985). Several factors affect salt accumulation in leaves: leaf age, shape,

Table 11.6 *Parameters for evaluation of commonly used irrigation methods in relation to risk reduction*

Evaluation parameter	Irrigation method			
	Furrow irrigation	Border irrigation	Sprinkler irrigation	Drip irrigation
Foliar wetting and consequent leaf damage resulting in poor yield	No foliar injury as the crop is planted on the ridge	Some bottom leaves may be affected but the damage is not so serious as to reduce yield	Severe leaf damage can occur resulting in significant yield loss	No foliar injury occurs under this method of irrigation
Root zone salt accumulation with repeated applications	Salts tend to accumulate in the ridge which could harm the crop	Salts move vertically downwards and are not likely to accumulate in the root zone	Salt movement is downwards and root zone is not likely to accumulate salts	Salt movement is radial along the direction of water movement. A salt wedge is formed between drip points
Ability to maintain high soil water potential	Plants may be subject to stress between irrigations	Plants may be subject to water stress between irrigations	Not possible to maintain high soil water potential throughout the growing season	Possible to maintain high soil water potential throughout the growing season and minimize the effect of salinity
Suitability to handle brackish wastewater without significant yield loss	Fair to medium. With good management and drainage acceptable yields are possible	Fair to medium. Good irrigation and drainage practices can produce acceptable yields	Poor to fair. Most crops suffer from leaf damage and yield is low	Excellent to good. Almost all crops can be grown with very little reduction in yield

Source: Adapted from Pescod (1992)

angle, and position on plant; type and concentration of salt; ambient temperature; air velocity; irrigation frequency; and length of time the leaf remains wet (Maas and Grattan, 1999). Since the problem is related more to the frequency than the duration of sprinkler irrigation, infrequent and heavy irrigations should be preferred over frequent and light irrigations (Qadir and Minhas, 2008). Several parameters for the evaluation of commonly used irrigation methods in relation to risk reduction are given in Table 11.6.

Irrigation, drainage, and root-zone salinity management

While using saline water or wastewater, the volume of irrigation water applied should be in excess of crop water requirement (evapotranspiration) and predictable rainfall should be taken into consideration as it leaches excess salts from the root zone. Salinity control by effective leaching of the root zone therefore becomes an important option for farmers who do not have limited water allocations. In order to calculate leaching requirement, farmers will need assistance to analyse the electrical conductivity of their soils and irrigation water so that the following equation can be used.

$$LR = EC_w / [5(EC_e) - (EC_w)] \qquad 11.1$$

LR refers to leaching requirement (additional water fraction of the irrigation water) needed to control salts in the root zone within the salt tolerance level of a specific crop with the routine surface irrigation method, i.e. the fraction of infiltrated water that must pass through the root zone to keep soil salinity within a specific level. EC_w is electrical conductivity of applied irrigation water expressed in terms of dS m^{-1}. EC_e refers to the average soil salinity (determined from the extract of saturated soil paste; also expressed as dS m^{-1}) in the root zone that can be tolerated by the crop under consideration. The values given in Table 11.5 for different crops can be used. These values also provide information on yield loss by these crops as the salinity of the growth medium increases.

The LR is needed to calculate the total water requirement (AW) of the crop. This can be estimated from Equation 11.2 (Ayers and Westcot, 1985).

$$AW = ET / (1 - LR) \qquad 11.2$$

AW refers to the depth of applied water per unit area on a yearly or seasonal basis (mm yr^{-1}); ET is the annual or seasonal crop water consumption expressed as evapotranspiration (mm yr^{-1}); and LR is the leaching requirement expressed as a fraction (see above). Both AW and ET can also be expressed in terms of m^3 of water (1mm = 10m^3 ha^{-1}).

The leaching required to maintain salt balance in the root zone may be achieved either by applying sufficient water at each irrigation to meet the LR or by applying, less frequently, a leaching irrigation sufficient to remove the salts accumulated from previous irrigations. The leaching frequency depends on the salinity status in water or soil, salt tolerance of the crop and climatic conditions (Qadir and Minhas, 2008). The amount of rainfall should be taken into consideration while estimating the leaching requirement and selecting the leaching method. Although leaching is essential to prevent root-zone salinity, leaching under saline wastewater irrigation may result in the movement of nitrates, metals, metalloids and salts to the groundwater. Therefore, monitoring of groundwater levels and quality is an essential indicator of environmental performance (Lazarova and Bahri, 2005).

Adequate soil drainage is considered to be an essential prerequisite to achieving leaching requirement vis-à-vis salinity control in the root zone. Natural internal drainage alone may be adequate if there is sufficient storage capacity in the soil profile or a permeable subsurface layer occurs that drains to a suitable outlet. An artificial system must be provided if such natural drainage is not present. Otherwise the resultant root-zone salinity control will not be sustainable. Besides, adequate soil drainage, land-levelling and adequate depth of groundwater are also basic components to maintain salinity in the root zone at a specific level. The suitable depth of groundwater depends on climate, groundwater quality and crop(s) to be grown.

Conjunctive use with freshwater

Saline wastewater can be used for irrigation in conjunction with freshwater, if available, through cyclic and blending approaches. Several studies have evaluated different aspects of these approaches on a field scale (Oster, 1994; Qadir and Oster, 2004; Rhoades, 1989; Sharma and Rao, 1998; Shennan et al., 1995). These approaches allow a good degree of flexibility to fit into different situations. Guidelines pertaining to water quality for irrigation in terms of salinity- and sodicity-related parameters were mentioned in Chapters 2 and 6 in this volume.

The cyclic strategy involves the use of saline wastewater and non-saline irrigation water in crop rotations that include both moderately salt-sensitive and salt-tolerant crops. Typically, the non-saline water is also used before planting and during initial growth stages of the salt-tolerant crop while saline water is usually used after seedling establishment (Oster, 1994; Rhoades, 1989). The cyclic strategy requires a crop-rotation plan that can make best use of the available good-quality water and saline wastewater, and takes into account the different salt sensitivities among the crops grown in the region, including the changes in salt sensitivities of crops at different stages of growth. The advantages of the cyclic strategy include:

- Steady-state salinity conditions in the soil profile are never reached because the quality of irrigation water changes over time.
- Soil salinity is kept lower over time, especially in the topsoil during seedling establishment.
- A broad range of crops, including those with high economic-value and moderate salt sensitivity, can be grown in rotation with salt-tolerant crops.
- Conventional irrigation systems can be used.

Studies addressing the cyclic use of drainage waters (Oster, 1994; Rhoades, 1989; Shennan et al., 1995) have shown that this strategy is sustainable for cotton, wheat, safflower (*Carthamus tinctorius* L.), sugar beet, tomato (*Lycopersicon esculentum* Mill.), cantaloupe (*Cucumis melo* L.) and pistachio (*Pistacia vera* L.), provided that the problems of crusting or poor aeration are dealt with through optimum management. Sharma and Rao (1998) provided further evidence from a study area where waters with various levels of salinity (EC = 6, 9, 12, 18.8 dS m^{-1}) were used successfully for seven years to irrigate different crops like wheat, pearl millet (*Pennisetum glaucum* (L.) R. Br.) and sorghum with acceptable yield reductions but without any serious degradation of a coarse-textured soil. The soil salinity levels were managed satisfactorily by monsoon rains and in part pre-sowing irrigation of 70mm with low-salinity canal water. However, the extent of salt leaching was heavily dependent on the total amount of monsoon rainfall and subsurface drainage.

Blending consists of mixing good- and poor-quality water supplies before or during irrigation. Saline wastewater can be pumped directly into the nearest irrigation canal or water channel. The quantity of saline wastewater pumped into the canal can be regulated so that target salinities in the blended water can be achieved (Oster, 1994; Rhoades, 1989). Water qualities are altered, according to the availability of different irrigation water qualities and quantities, between or within an irrigation event. Blending saline waters with good-quality irrigation waters has been a common practice in several countries such as India, Pakistan and the USA (Minhas, 1996; Qadir and Oster, 2004).

Seedbed preparation and planting techniques

Since most crops are salt-sensitive at germination stage, it is important to avoid the use of saline wastewater at this critical time. Under field conditions, it is possible, by modifications of planting practices, to minimize salt-accumulation around the seed and to improve the standing of crops that are sensitive to salts during germination. These modifications can include sowing near the bottom of the furrows on both sides of the ridges, raising seedlings with freshwater and their transplanting, using mulches to carry over soil moisture for longer period and increasing the seed or seedling rate per unit area (plant density) to compensate for possible decrease in germination and growth (Minhas, 1996; Tanji and Kielen, 2002).

Soil and water treatment

Irrigation with sodic wastewater needs provision of a source of Ca^{2+} to mitigate Na^+ effects on soils and crops. Gypsum ($CaSO_4{\cdot}2H_2O$) is the most commonly used source of Ca^{2+}; its requirement for sodic water depends on the Na^+ concentration and can be estimated through simple analytical tests. Gypsum can be added to the soil, applied with irrigation water by using gypsum beds or placing gypsum stones in water channels. In the case of calcareous soils containing precipitated or native calcite ($CaCO_3$), none or a much lower rate of gypsum application may work well. Plant residues and other organic matter left in or added to the field can also improve the chemical and physical conditions of soils irrigated with sodic wastewater. In addition, biological treatment of salt-prone wastewater by standard activated sludge culture can be triggered by the inclusion of salt-tolerant organisms to improve treatment efficiency.

Where available, high-electrolyte waters containing an adequate proportion of divalent cations such as Ca^{2+} can be used for sodic and saline-sodic soil amelioration. These waters can improve soil hydraulic properties without the need to apply a calcium-supplying amendment (Qadir et al., 2007a; Quirk, 2001). However, the ratio of divalent cations, particularly Ca^{2+}, to total cations (TC) in the applied water should be at least 0.3. Synthesis of the data on total cationic and Ca^{2+} concentrations in several wastewater samples suggests that wastewaters have a wide range of calcium to TC ratio (C_{Ca}:C_{TC}), i.e. from as low as 0.03 to as high as 0.80 (Table 11.7). These contrasting observations reveal that the use of wastewater to irrigate sodic soils should be carefully planned as the C_{Ca}:C_{TC} should be over the threshold value of 0.3. Several studies have demonstrated that adequate amounts of Ca^{2+} supplied through irrigation water or applied to the soil in the form of some amendment improve soil structure and counterbalance the negative effects of high concentrations of Na^+ when sodic soils are brought under cultivation (Oster et al., 1999; Qadir et al., 2001).

The applicability of the high-electrolyte water is effective under certain conditions:

- The sodic soil under amelioration and management has smectite- and montmorillonite-type clay minerals with low hydraulic conductivity.
- The soil physical condition has deteriorated and hydraulic conductivity is so low that the time required for amelioration or the amount of amendment required is excessive.
- The irrigation water to be used following amelioration is so low in electrolyte concentration that water transmission would decrease adversely.

Table 11.7 *Concentrations of total cations (mmol$_c$ per litre) and calcium (mmol$_c$ per litre), and ratio of calcium to total cations in wastewater samples*

Total cations (C_{TC})[a]	Calcium (C_{Ca})	C_{Ca} : C_{TC}	Reference
7.0	1.6	0.23	Kaul et al. (2002)
10.0	2.7	0.27	Kaul et al. (2002)
17.0	3.7	0.22	Mitra and Gupta (1999)
19.0	5.0	0.26	Mitra and Gupta (1999)
8.0	2.5	0.31	Arora et al. (1985)
9.0	2.8	0.31	Baddesha et al. (1986)
9.0	7.2	0.80	CSSRI (2004)
21.0	11.0	0.52	CSSRI (2004)
44.0	1.5	0.03	Ensink et al. (2002)

[a] $C_{TC} \approx$ EC (dS m^{-1}) x 10.

ORGANIC CONTAMINANTS

Exposure of consumers, farmers and crops in developing countries to organic contaminants is probably much higher through direct pesticide application than via contaminated irrigation water. The challenge of any related risk (and its mitigation) starts with its assessment, which is costly if based on actual analysis (see Chapter 6). A possible alternative for pesticides is to predict the risk based on easier to measure environmental factors and application practices, using, for example, the free Pesticide Impact Rating Index (PIRI) software, mentioned in Chapter 6, which was developed in Australia but also been applied elsewhere, like Sri Lanka. More difficult and costly would be the analysis of organic contaminants of emerging concern, like residual pharmaceuticals or endocrine disruptor compounds. This limits the current knowledge on their actual risk in wastewater irrigation, which has so far been ranked as relatively low compared, for example, to pathogenic hazards (Chang et al., 2002; Toze, 2006b; WHO, 2006b).

To address organic contaminants preventive measures are therefore more suitable than any soil or water treatment. Key activities include the use of alternative pesticides or integrated pest management. In order to avoid pesticides entering streams used for irrigation or other purposes, buffer zones, run-off reduction and the use of wetlands for remediation could be considered. Containment of contaminated water in dams or wetlands may provide time for pesticides to be removed by sediments or through degradation. Farming practices that reduce run-off, such as the provision of vegetation cover or vegetated bufferstrips (Box 11.3), can significantly reduce the probability of environmental impacts (Finlayson and Silburn, 1996; Kennedy, 1999; USDA, 2000). In spiking trials, the FILTER system has also been shown to reduce pesticide loads by more than 98 per cent (Biswas et al., 2000).

The key removal mechanisms for most organic substances are adsorption and biodegradation in soils and sediments (WHO, 2006b). Removal efficiencies are greater in soils rich in silt, clay and organic matter. Black carbon, in particular, can play a significant role in fixing highly toxic polycyclic aromatic hydrocarbons, polychlorinated biphenyls, dioxins, polybrominated diphenylethers and pesticides (Koelmans et al., 2006).

Chemical stability and slow natural attenuation of certain POPs, such as polychlorinated biphenyls (PCBs) and 1,1,1-trichloro-2,2-bis(4-chlorophenyl) ethane (DDT), make remediation of these compounds a particularly intractable environmental challenge. The approach usually taken is to isolate affected sites and either remove the contaminated soil or rely on phytoremediation as described above.

BOX 11.3 BUFFER-STRIPS

There is a dearth of empirical evidence on the performance of various options for mitigating diffuse pollution from agriculture. Especially, riparian buffers have received significant attention over the past 20 years. Ranges for positive buffer efficacy were found to be 30–100 per cent for soil sediment, 30–95 per cent for total phosphorus, 10–100 per cent for total nitrogen, 30–100 per cent for pesticides and 53–100 per cent for faecal indicator organisms. Since many of the experiments underpinning these data were conducted under ‘ideal’ operating conditions, it is likely that buffer performance in nature will be lower. Overall, the evidence base suggests that buffers provide at least useful short-term benefits, while longer-term impacts remain questionable owing to risks of pollution swapping (Collins et al., 2009).

CONCLUSIONS

There is a variety of management options for smallholder farmers in developing countries to address the challenges and risks of exposure to heavy metals or excessive salts and nutrients through irrigation water. These measures include soil- and water-based interventions as well as changes in crops and crop varieties. Currently available techniques that have been successfully applied to remediate metal or metalloid contaminated soils include in situ and ex situ engineering options, irrigation management options, in situ soil-based immobilization, phytoremediation, chelate-enhanced phytoextraction, etc. In certain cases, farmers and authorities might have no other choice than to cultivate better adapted and non-edible crops, or to zone the areas for non-agricultural land use. In view of possible organic contaminants, appropriate pest and pesticide management will remain more important than soil and water treatment. All methods have however also their drawbacks in effectiveness, duration and economics (Iskandar and Adriano, 1997; Zaurov et al., 1999). Due to the additional risk of bioaccumulation it is in many cases not possible to provide details on the general effectiveness of measures in terms of health-risk reduction, which will largely depend on a variety of site conditions, as well as spatial and temporal factors. While our knowledge is much advanced in view of challenges related to excess nutrients and salts, large gaps remain for heavy metals and, in particular, organic contaminants. A key constraint to risk assessments and mitigation is the missing capacity to analyse and monitor these constituents, especially in developing countries. It remains, therefore, crucial to support pollution preventing policies and measures, including the reduction of possible fertilizer subsidies where they have led to over-fertilization. In the case of metals, metalloids, nutrients and emerging contaminants, pre-treatment and/or segregation of industrial wastewater from the domestic and municipal wastewater stream (eventually used for irrigation) should have highest priority

(Patwardhan, 2008). Also, the sources of salts in wastewater can be reduced by using technologies in the industrial sector that reduce salt consumption vis-à-vis discharge into the sewage system. In addition, many hazardous chemicals can be replaced in production processes and restrictions can be imposed on the use of certain products for domestic use that are major sources of, for example, salts in wastewater (Lazarova and Bahri, 2005).

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