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Assessing the State of Knowledge and Impacts of Recycled Water Irrigation on Agricultural Crops and Soils

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Assessing the State of Knowledge and Impacts of Recycled Water Irrigation on Agricultural Crops and Soils

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Assessing the State of Knowledge and Impacts of Recycled Water Irrigation on Agricultural Crops and Soils

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Acronyms and Abbreviations

AOP	Advanced Oxidation Process
BAR	Bioaccumulation factor
BCF	Bioconcentration factor
BFRs	Brominated flame retardants
BR	Blending ratio
CEC	Contaminants of emerging concern
CROSS	Cation ratio of structural stability
DDI	Daily dietary index
DIM	Daily Intake of Metal
DPU	Dynamic plant uptake
DU	Distribution uniformity
EC_{50}	Average rootzone soil salinity where 50% yield loss is predicted
ECe	Average rootzone salinity of the saturated paste soil extract (dS/m)
ECw	Electrical conductivity of irrigation water (dS/m)
EDI	Estimated daily intake
EHEC	Enterohemorrhagic
EPS	Exopolysaccharides
ESP	Exchangeable sodium percentage
ET	Evapotranspiration
Fc	Concentration factor
GAC	Granular Activated Carbon
GAP	Good Agricultural Practices
HI	Hazard Index
HQ	Hazard quotient
HRI	Health Risks Index
Kc	Crop coefficient
LF	Leaching fraction
LR	Leaching Requirement
PAC	Powdered activated carbon
PBDEs	Polybrominated diphenyl ethers
PFASs	Polyfluoroalkyl substances
PFOS	Perfluorooctane sulfonate
PPCPs	Pharmaceuticals and personal care products
QMEAs	Quantitative Microbial Exposure Assessments
QMEAs	Quantitative Microbial Exposure Assessments
QMRA	Quantitative Microbial Health Risk Assessment
QTL	Quantitative trait locus

RCF	Root concentration factor
RO	Reverse osmosis
s	Slope (% yield decline per dS/m)
SAR	Sodium adsorption ratio
STEC	Shiga-toxin producing *E. coli*
t	Salinity threshold
TDS	Total dissolved solids (mg/L)
THQ	Target Hazard Quotient
TSCF	Transpiration stream concentration factor
TTC	Threshold of toxicological concern
TWW	Treated wastewater
WWTP	Wastewater treatment plant
Yr	Relative yield

Executive Summary

ES.1 Key Findings

- New cultivars are needed, and development will be supported by further research into salt tolerant under field conditions and to better characterize plant response to salinity in heterogeneous soil conditions, particularly under microirrigation as salinity in the soil water (ECsw) is continuously changing over space and time.
- Management practices such as blending, cycling, and sequential use should be adopted when saline-sodic recycled water is used for irrigation.
- The quality of the recycled water can contribute to the number of heavy metals in agricultural soils affecting the microbiological balance of soils and reducing soil fertility.
- When agricultural fields are irrigated with recycled water, constituents of emerging concern (CECs) are unlikely to significantly accumulate in the soil, as most CECs are susceptible to degradation in multiple pathways. However, due to the incapacity to evaluate the cocktail effect of CECs, as well as lack of knowledge regarding the toxicity of CEC transformation products, the actual risk may be underestimated.
- To date, there is little evidence to suggest that adequately treated recycled water poses more risk in terms of waterborne microbial pathogens for produce-related illness or outbreaks than other sources of irrigation water, but epidemiological and quantitative risk assessment models suggest that guidelines for the use of recycled water should be regionally specific and consider overall population health.

ES.2 Background and Objectives

Population growth, rapid urbanization, and climate change have been contributing to water scarcity in many regions in the world. Access to adequate and safe freshwater is one of the grand challenges of this time (Sheidaei et al., 2016). Accounting for 70% of global freshwater withdrawals, agriculture is suffering the greatest impact from the water shortage (Norton-Brandão et al., 2013; FAO, 2017). To relieve the pressure on water supplies, municipal-treated wastewater (referred to as recycled water here forth) has been recognized as an important alternative source for irrigation water and is increasingly being applied in arid and semi-arid regions (Hamilton et al., 2007; Qadir et al., 2010; Grattan et al., 2015; Otoo and Drechsel, 2018). In California, about 46% of treated wastewater is recycled for agricultural use, while in Florida, the fraction accounts for 44% (Bryck et al., 2008). In China, recycled water irrigation began in 1957 and the reclamation rate of treated wastewater increased to 62% in 2014 in the cities that pioneered the implementation of wastewater reclamation and reuse (Wang et al., 2017; Zhang et al., 2018). Wastewater reuse has been long practiced in the Mediterranean basin, especially in the more water-scarce regions where the treated wastewater reuse is up to 5-12% of the total amount of treated wastewater effluent (Rygaard et al., 2011; Agrafioti et al., 2012; Kathijotes and Panayiotou, 2013; Kellis et al., 2013; Navarro et al., 2018; Saliba et al., 2018). Overall, GIS-based analysis has shown that the land area irrigated with recycled water increased from 20 million hectares in 2007 to 36 million hectares in 2017, which represents approximately 10% of the world irrigation area (Hamilton et al., 2007; Thebo et al., 2017). The reuse of treated wastewater offers many potential benefits, such as 1) decreasing stress on freshwater supply; 2) reducing cost and energy consumption (Meneses et al., 2010); 3) recycling nutrients and helping maintain soil fertility (Hanjra et al., 2012; Becerra-Castro et al., 2015; Hassena et al., 2018;); 4) reducing discharge from sewage treatment plants into the environment (Meneses et al., 2010; Plumlee et al., 2012); and 5) avoiding the impact of new water supply developments (e.g., dams, reservoirs).

Impacts of recycled water used for irrigation of agricultural lands are generally voiced and listed in the following categories:

- Reduction of the yield of crops due to the higher salt levels
- Injury to crops and ornamentals from specific elements (e.g., sodium, and boron)
- Degradation of soil structure in the long term due to higher sodium levels—or lower calcium and magnesium concentrations
- Degradation of groundwater quality as a result of leachates from the root zone, ultimately arriving at the water table and mixing with ambient water in an unconfined aquifer underlying the recycled-water-irrigated lands
- Uptake of CECs into the edible tissues of plants and detected levels of those compounds in humans consuming crops grown with recycled water
- Higher costs imposed on utilities due to the higher treatment levels that may be required to mitigate some of the above impacts
- Increased yield of some crops due to higher levels of nutrients in recycled water, thereby reducing fertilizer requirement

The overall research objective of this project is to assess the state of knowledge and impacts of recycled water irrigation on agricultural crops.

ES.3 Project Approach

The research team used a three-pronged approach to assess the state of knowledge and impacts of recycled water irrigation on agricultural crops.

- First, research conducted to date was reviewed and summarized, highlighting conditions under which significant impacts have been reported. Classical texts and contemporary literature on recycled water reuse for irrigation were reviewed.
- Second, the team worked closely with the project partners (utilities that supply recycled water for irrigation [Monterey One and Pajaro Valley Water Management Agency]) and access their water quality characteristics, farming patterns, and farmers' responses to the use of recycled water.
- Third, the research team collaborated with researchers from other countries that use recycled water for agricultural irrigation to collate data and information from their experiences with using recycled water for irrigation. This was done in form of select case studies in Australia, Israel, Spain, and Chile.
- A draft final report was prepared for review by the internal QA/QC team before submission to The Water Research Foundation.

ES.4 Results

More research is needed to develop cultivars that are more tolerant in field conditions and to better characterize plant response to salinity in heterogeneous soil conditions, particularly as drip and other low-pressure irrigation systems become more and more prevalent. This new information is critical as recycled water produced by various technologies continues to expand in arid and semi-arid climates.

The use of recycled water high in sodium and potassium can adversely affect soil in the form of reduced infiltration, poor soil tilth, and poor aeration resulting in anoxic conditions in the root zone. These negative impacts can be minimized with amendments like gypsum, sulfur, and sulfuric acid. Management practices such as blending, cycling, and sequential use should be adopted when saline-sodic recycled water is used.

The quality of the recycled water can contribute to the number of heavy metals in agricultural soils affecting the microbiological balance of soils and reducing soil fertility. Such impact can negatively soil health.

When agricultural fields are irrigated with recycled water, CECs are unlikely to significantly accumulate in the soil, as most CECs are susceptible to degradation in multiple pathways. Studies to date have suggested that CECs introduced into the soil via irrigation are mainly accumulated in the surface soil layer; only CECs with low sorption capacity and long persistence may be leached appreciably under intensive or long-term irrigation. However, due to the incapacity to evaluate the cocktail effect of CECs as well as lack of knowledge regarding the toxicity of CEC transformation products, the actual risk may be underestimated. More research is urgently needed to fill these knowledge gaps to better elucidate the fate and risks of trace-level CECs in the recycled water irrigation-soil-plant-human continuum and ultimately the exposure to humans via dietary intakes of the impacted agricultural products, as well as the ecological risk of CECs toward non-target terrestrial organisms.

To date, there is little evidence to suggest that adequately treated recycled water poses more risk in terms of waterborne microbial pathogens for produce-related illness or outbreaks than other sources of irrigation water, but epidemiological and quantitative risk assessment models suggest that guidelines for the use of recycled water should be regionally specific and consider overall population health.

Strict regulations and successful case studies have helped to build public acceptance of recycled water reuse for irrigation in California, Arizona, Texas, and Florida. Other countries such as Australia, Chile, Israel, and Spain have also developed successful recycled water reuse projects.

ES.5 Benefits

The project findings fill critical knowledge gaps on the impact of recycled water reuse on soil and crop productivity. Specifically, utilities, farmers, and policymakers will find information on the potential impact of recycled water salinity and sodicity important. Utilizers and policymakers will also find information on CECs and heavy metals relevant to their operation. For example, the finding that because agricultural fields are irrigated with recycled water, CECs are unlikely to significantly accumulate in the soil, as most CECs are susceptible to degradation in multiple pathways, will be useful to the utilities that supply recycled water to growers. Current research also indicates the risk from waterborne microbial pathogens for produce-crop is not different from that of crops irrigated with freshwater.

ES.6 Related WRF Research

- Addressing Impediments and Incentives for Agricultural Reuse (4956)
- Evaluating Economic and Environmental Benefits of Water Reuse for Agriculture (4829)
- Agricultural Reuse-Impediments and Incentives (4775)

CHAPTER 1

Crop Salinity Tolerance and Ion Toxicity

1.1 Introduction

Treated municipal wastewaters contain mineral salts, but the concentration and composition of these salts vary widely among locations and sources of water (Ayers and Westcot, 1985; Wallender and Tanji, 2012). These salts dissolve in solution to form ions (cations and anions) where the most common cations are calcium (Ca^{2+}), magnesium (Mg^{2+}), and sodium (Na^{+}), and the most abundant anions are chloride (Cl^{-}), sulfate ($SO4^{2-}$) and bicarbonate (HCO_3^{-}). Potassium (K^{+}), carbonate (CO_3^{2-}), nitrate (NO_3^{-}), phosphate ($H_2PO_4^{-}$), boron (B), and trace elements also exist in soils and water supplies but most often their concentrations are comparatively low even though their presence can still influence crop growth and management. In rare instances, wastewater may contain heavy metals but typically these, if present, are concentrated in sewage sludge and are not problematic to the use of recycled water. The overall concentration of these constituents reflects the overall salinity of the water which can be characterized different ways.

The salinity of the irrigation water is usually expressed by its electrical conductivity (ECw) because the salts dissolved in the water form ions and conduct electrical current (USDA-USSL, 1954). The standard unit of ECw is decisiemens per meter (dS/m), which is numerically equivalent to millimhos per centimeter (mmho/cm). The EC of water is readily measured using a conductivity meter standardized to its reading at 25 ^{o}C (USDA-USSL Staff, 1954).

Salinity is also expressed as total dissolved solids (TDS) with units reported in mg/L which is, for all practical purposes, numerically equivalent to parts per million (ppm). TDS represents the mass of salt that remains after a liter of water is evaporated to dryness. This term is still reported by many analytical laboratories and is used widely by wastewater engineers.

The salinity parameters ECw and TDS are, for the most part, linearly related to one another over the concentration range where most crops are impacted and where most waste waters are used for irrigation. The most common conversion is TDS = 640 EC (dS/m) (USDA-USSL Staff, 1954) but this conversion is dependent upon the composition and concentration of the water. As such, Rhoades et al (1992) suggest an approximate relationship of water as ECw of 1 dS/m = 10 mmho/L = 700 mg/L. For ECw > 5 dS/m, Hanson et al. (2006) suggest that the conversion TDS (mg/L) = 800 EC (dS/m) is more accurate. The chloride concentration in water also serves as a regulatory parameter to water salinity in many cases since chloride is usually the most abundant anion in most water resources and its concentration relates to salinity.

Irrigation water supplies that are low in salinity are in limited supply across the globe, particularly in arid and semi-arid climates. Therefore, waters of poorer quality will be used more and more in the future to satisfy crop water needs including municipal wastewater. And, with a changing climate and uncertainty in precipitation patterns, wastewaters will likely play a larger role supplementing irrigation water supplies. However, due to their higher concentration of salts, wastewaters present challenges for sustained long-term use in irrigated agriculture.

1.1.1 Recycled Desalinated Water

Desalination of saline waters has increased over the years and desalinated water has been used for irrigation. Desalinated water, originating from the sea, saline aquifers or wastewater treatment plant (WWTP) effluents, plus additional constituents resulting from its municipal use, eventually dictate the quality of the wastewater generated by the water consumer. This modified quality in turn impacts the processes at the treatment plant. Ultimately, upon irrigation, the quality of treated effluent the WWTP produces can have significant effects on the crop, soil conditions, and the surrounding environment including groundwater. A schematic, holistic flowsheet of desalinated seawater and its use in irrigated agriculture is depicted in Figure 1-1.

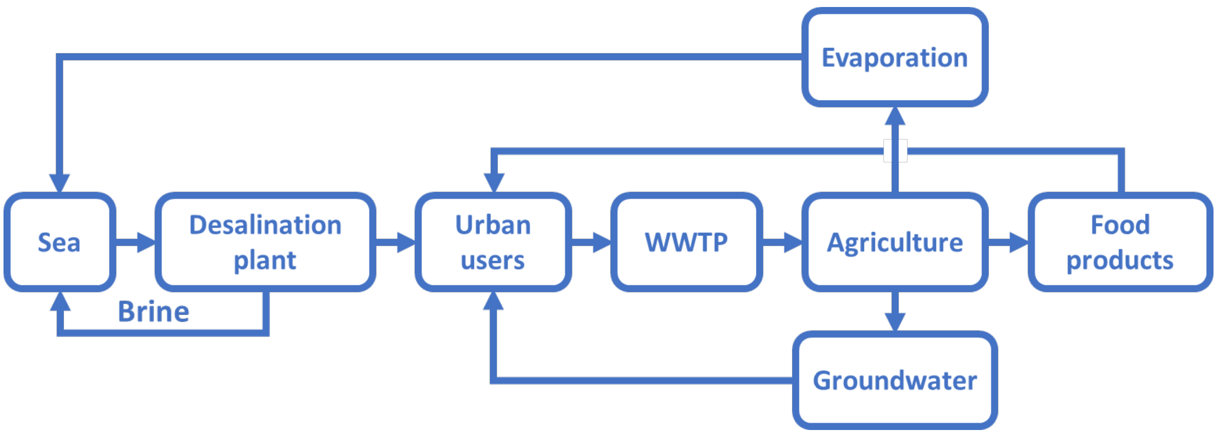

Figure 1-1. Holistic Interpretation of the Desalination – Urban-WWTP-Agriculture Water Use Pathway.

Desalination has a profound influence on the quality of the recycled water used for irrigation. The quality can be beneficial or detrimental. On one hand, adding desalinated water to irrigation networks reduces water salinity and the salt load which is quite favorable in most agricultural lands. On the other hand, excess constituents such as boron, which often passes through RO (reverse osmosis) membranes, or lack of minerals such as magnesium and sulfate, which are completely removed by the SWRO (seawater RO) plant process, may have detrimental effects on the crop, mineral nutrient relations or soil physical conditions (Penn et al., 2009, Lahav et al., 2010). Similarly, various contaminants in the effluents that are not removed by treatment or affected by transport processes in the vadose zone may reach natural aquifers and have long term negative consequences.

1.1.2 Irrigation Water Quality Guidelines

General water quality guidelines have been developed to assist water users in assessing the suitability of water for irrigation. The US EPA (2012) has generally adopted the agricultural water quality guidelines presented by Ayers and Westcot (1985) in their FAO 29 publication entitled “Water Quality for Agriculture”. These guidelines are presented in Table 1-1 and should only be viewed as a first approximation of water quality concerns as crop tolerance and irrigation management can also influence the ‘degree of restriction of use’. The category ‘none’ includes concentrations or values at or below this level that typically would not be problematic for the vast majority of crops across the globe, representing a variety of climates, where other parameters or factors are not restricting crop production. Categories of 'slight to moderate' and 'severe' suggest that concentrations or values within this range may impose various restrictions depending upon the crop type, management, and environmental conditions.

Table 1-1. Irrigation Water Quality Parameters in Wastewater and Guideline Concentrations Potentially Affecting the Degree of Restriction of Use in Irrigated Agriculture.

Source: USEPA, 2012a.

Degree of Restriction on Use				
Parameters	**Units**	**None**	**Slight to Moderate**	**Severe**
Salinity				
ECw	dS m^{-1}	< 0.7	0.7 - 3.0	> 3.0
TDS	mg/L	< 450	450 - 2000	> 2000
Ion Toxicity	SAR	< 3	3 - 9	> 9
Sodium (Na)	meg/L	< 3	> 3	
Root Absorption	mg/L	< 70	> 70	
Foliar Absorption	meg/L	< 2	2 - 10	> 10
Chloride (Cl)	mg/L	< 70	70 - 355	> 355
Root Absorption	meg/L	< 3	> 3	
Foliar Absorption	mg/L	< 100	> 100	
Boron	mg/L	< 1.0	1.0 - 2.0	> 2.0
pH			6.5 - 8.4	

The inorganic composition of wastewaters used for irrigation in many of the leading countries utilizing wastewater use can vary widely from location to location. The composition of the recycled water is largely dependent upon the quality of the water used before it is sent to the sewage treatment plant. But the ultimate salt concentration is always higher than the source water provided the treated water is not desalinized prior to reuse. In Israel, for example, the contribution of additional chloride to wastewater by domestic use averages about 100 mg/L. Table 1-2 represents the salt composition and the sodium adsorption ratio (SAR) for various recycled waters across the globe.

Treated municipal wastewater quality data from these countries indicate that salinity, as indicated by the electrical conductivity of the water (EC), ranges between 0.9 to 3.0 dS/m which can be problematic for most sensitive to even moderately salt-tolerant crops. Specific ions (Na+, Cl- and B) are also elevated and may pose injury to sensitive crops, particularly tree crops grafted on salt-sensitive rootstocks. Sodium and the SAR of the water may pose problems in terms of sodium toxicity to susceptible crops or may cause soil structural degradation leading to poor infiltration without proper management.

Table 1-2. Typical Concentrations of Critical Inorganic Water Quality Constituents in Treated Municipal Wastewater from Several Countries with Mediterranean Climates Used for Irrigated Agriculture.

Inorganic Constituent	Units	Israel (Southern coastal plain)	Italy (Puglia)	Spain (Murcia)	USA (Monterey, Pajaro Valley California)	Australia
Cl	mg/L	202- 326	161-460	200-764	118-325	412-550
Na	mg/L	140- 203	101-328	146-435	114-214	285-334
SAR	-	3.8- 5.1	0.9-6.5	2.3-8.3	3.0-5.3	8.1-11.1
B	mg/L	0.2	0.1-1.0	0.1-0.7	0.3-0.4	0.2-0.3
EC	dS/m	1.3- 1.7	0.9-2.9	1.2-3.0	1.2-1.9	1.5-2.0
References		Aharoni et al., 2018; Erel et al., 2019; Yasuor et al., 2020.	Lopez et al., 2006; Palese et al., 2006; Triqueros et al., 2019; Vivaldi et al., 2019; Pedrero et al., 2019	Maestre-Valero et al., 2019	Monterey 1 Water, 2019; Pajaro Valley Water, 2019; Sheikh et al., 1990	Awad et al., 2019; Barwon Water, 2020

1.2 Past and Current Knowledge

Salinity has impacted irrigated agriculture for thousands of years. In ancient Mesopotamia, the Fertile Crescent was inundated by salts due to inadequate drainage which led to the destruction of this ancient hydraulic-based civilization (Hillel, 2000). But salinity was not unique to the middle east. In the early part of the 20th century, it was recognized by the United States Department of Agriculture (USDA) that agricultural production in the western part of the United States was being impacted by salinity. To address this concern, the USDA Salinity Laboratory was constructed in Riverside, California shortly after World War II to study the effects of salinity on soils and crop production. The Salinity Laboratory published the famous USDA Handbook 60 entitled "Saline and alkali soils" in 1954 to help with the diagnosis and improvement of saline and alkali soils (USDA-USSL, 1954) and continues to be a reference cited by salinity researchers across the globe.

The US Salinity Laboratory made a distinction between soils that were saline and alkaline. Saline soils were those with an electrical conductivity of the saturated soil paste > 4 dS/m while alkaline soils with those having an exchangeable sodium percentage (ESP) > 15 (USDA-USSL, 1954). Therefore, there are four general categories: 1) Non-saline, non-alkaline 2) Non-saline, alkaline, 3) Saline, non-alkaline and 4) Saline, alkaline. Because 'alkaline' refers to soils with above neutral pH, typically above 7.5, the term 'sodic' soil has replaced 'alkaline' to be more inclusive of soils across a larger pH range (Wallender and Tanji, 2012). Sodic soils and their impact on soil physical conditions will be discussed in a later section.

It has been recognized for decades that crops vary widely in their tolerance to salinity (USDA-USSL, 1954) as well as the basic physiological responses that account for these differences (Bernstein and Hayward, 1958). They understood that crop growth was impacted by osmotic inhibition of water absorption for the soil solution and by ion specific effects. These processes are not entirely independent upon one another and often impact the crop collectively.

Salinity reduces the osmotic potential of the soil solution thereby requiring the plant to osmotically adjust by concentrating solutes (i.e., ions or organic solutes) inside their cells in order to readily extract the water via osmosis. This concentration process requires metabolic energy (ATP) but its ultimate cost to plant growth depends on ion transport efficiencies across membranes and energy requirements to synthesize organic solutes, which differs among species and varieties within a species (Munns et al., 2020). The more energy used for these processes, the less is available for plant growth. As such, the efficiency of transport processes involving specific ions (e.g., Na^+) will affect the overall osmotic response. As a result, salt-stressed plants are stunted, even though they may appear healthy in all other regards (Bernstein, 1975). Both processes of adjustment (accumulation of ions and synthesis of organic solutes) occur but the extent to which one process dominates over the other is dependent upon the crop type and level of salinity (Läuchli and Grattan, 2012). And within the cell, compartmentalization is critical to keep toxic ions away from sensitive metabolic processes in the cytoplasm (Munns and Tester, 2008; Hasegawa et al., 2000). Such compartmentation is controlled by transport processes across the plasma membrane (i.e., cell membrane) and tonoplast (i.e. vacuolar membrane).

Specific ion effects can be directly toxic to the crop, due to excess accumulation of Na^+, Cl- or B in its tissue, or they may cause nutritional imbalances (Grieve et al., 2012). While specific ions reduce the osmotic potential of the soil solution, ion toxicities are rarely observed in annual crops grown in the field (with the exception of certain beans and soybeans) provided the ion ratios (e.g. Na^+/Ca^{2+}; Cl^-/SO_4^{2-}) are not extreme or salinity is not too high. However, when Na^+ dominates the cations or Cl^- concentrations are sufficiently high, these constituents can accumulate in older leaves and produce injury. Specific ion toxicities are particularly prominent in tree and vine crops and injury becomes more prevalent over the years. But rootstock selection plays a major role in controlling the amount of Na^+, Cl^-, and boron that accumulates in the scion (i.e., the variety grafted upon the rootstock) and thus their tolerance to these specific ions (Grieve et al, 2012). For example, in grapes, some rootstocks can differ in the transport of Cl^- to their leaves by as much as 15-fold (Bernstein, 1975). Specific ions can also induce nutritional disorders due to their effect on nutrient availability, competitive uptake, transport, and partitioning within the plant (Grattan and Grieve, 1999). For example, excess Na^+ can cause a sodium-induced Ca^{2+} or K^+ deficiency in many crops (Bernstein, 1975). These effects may be more subtle than direct ion toxicities but nonetheless affect the crop's performance. Specific-ion effects are addressed in more detail later in this section.

While osmotic and specific ion effects can occur concurrently, typically osmotic effects occur at early times while specific ion effects occur later (Munns and Tester, 2008). In the field, Na^+ and Cl^- toxicities can be observed in salt-affected fields after several years of tree or vine growth. Often Cl^- toxicity occurs in tree crops sooner than Na^+ toxicity as Na^+, unlike Cl^-, is retained in woody tissue, only to be released when sapwood converts to heartwood (Bernstein, 1975). The mechanisms of boron toxicity, on the other hand, are largely unknown but the most sensitive crops to boron tend to be those classified as boron mobile plants (e.g. almonds, plums, peaches, grapes). B-mobile species translocate B via polyols to growing tips where B toxicity is often manifested on trees as twig die back (Brown and Shelp, 1997).

Nutrient content on recycled water must be considered on the fertilization plan as most fertilizer materials are highly soluble salts, which dissociate in the soil solution following application. Seedling injury caused by fertilizer burn can result in minimal to extensive stand loss and can be extremely costly in high value vegetable crops. Soil conditions (texture, CE) are important for determining salt injury. Salt index (SI) of a fertilizer is a measure of the salt concentration that fertilizer induces in the soil solution (Mortvedt, 2001), so it is important to understand salt index and factors which contribute to fertilizer burn in order to avoid fertilizer injury to seedlings.

1.2.1 Expressions That Characterize Crop Salt-Tolerance

Rootzone salinity has traditionally been characterized by the electrical conductivity of the saturated soil paste (ECe) (USDA-USSL, 1954). While other methods such as 1:1; 1:2.5 and 1:5 extract ratios are quantitatively more reproducible and under Cl^- dominated conditions have shown good correlations with the chemistry in the saturated paste (Sonmez et al., 2008), the US Salinity Laboratory promoted the later method because 1) the chemistry of the saturated soil extract is close to that of the soil water and 2) the chemistry could nonetheless vary due to dissolution and precipitation of sulfate and carbonate minerals should larger soil water dilutions be used.

Because crops vary in their tolerance to salinity, scientists found it necessary to characterize their salt tolerance by developing simplistic models to predict their relative yield in the field as a function of seasonal average root zone salinity. The most comprehensive approach was performed in the 1970s by scientists at the US Salinity Laboratory (Maas and Hoffman, 1977). They collected and analyzed research papers describing salinity studies on a wide range of crops. When comparing studies, they understood, as did their predecessors (USDA-USSL Staff, 1954), that using absolute yield (mass/area) was an unreliable parameter to compare different crops types grown under a range of different conditions. Rather, the salt tolerance of crops can better be defined as a function of relative yield decline across a range of salt concentrations. Maas and Hoffman (1977) found that salt tolerance can be adequately measured on the basis of two parameters: 1) a “threshold” parameter which is the maximum root zone salinity (described as the electrical conductivity of the saturated soil extract, ECe) that the crop can tolerate above which yields decline and 2) the “slope” which describes the rate by which yields decline with increased soil salinity beyond the ‘threshold’ (Figure 1-2). Slope is simply the percentage of expected yield reduction per unit increase in salinity above the threshold value.

For soil salinities exceeding the threshold of any given crop, relative yield (Yr) or "yield potential" can be estimated using the following expression:

$$\text{Yield (\%)} = 100 - s(\text{ECe} - t) \qquad \text{(Equation 1-1)}$$

Where t = the ‘salinity threshold’ soil salinity value expressed in dS/m; s = the ‘slope’ expressed in % yield decline per dS/m; and ECe = average rootzone salinity of the saturated soil extract. The most current up-to-date listing of specific values for "t" and "s", called “salinity coefficients”, are found in a book chapter by Grieve et al. (2012) and are presented in Apendix 1. The greater the threshold value and lower the slope, the greater the salt tolerance.

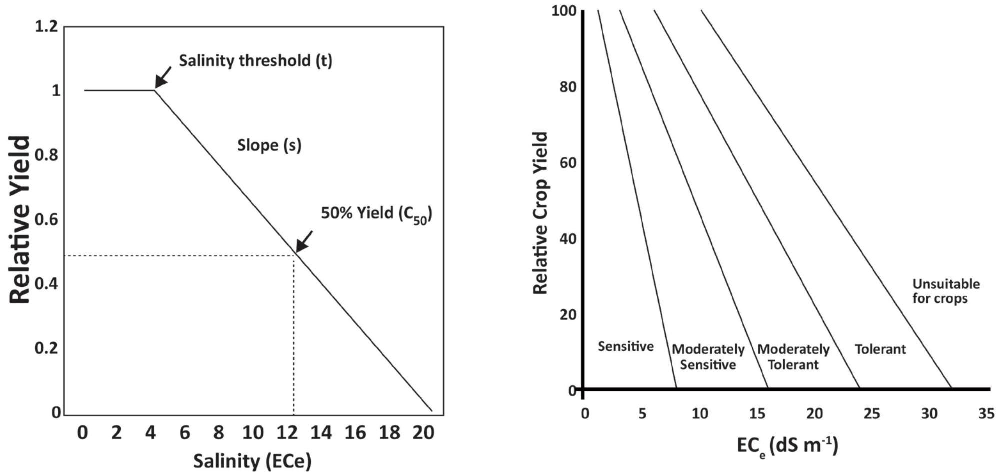

Figure 1-2. Salt Tolerance Parameters 'Salinity Threshold' (t) and Slope of Yield Decline (s) for Salinity That Exceeds the 'Threshold' (left) and Salt Tolerance Categories First Described by Maas and Hoffman, 1977 (right).
Source: (Left) Reprinted from *Scientia Horticulturae* 78 (1998); by Shannon, M.C., and C. M. Grieve; "Tolerance of Vegetable Crops to Salinity"; pp. 5-38; Copyright (1998), with permission from Elsevier.
(Right): Adapted from Maas and Hoffman, 1977.

Most agronomic grain crops such as barley, oats, rye and wheat are much more tolerant to salinity than most horticultural tree and vine crops such as almond, berries, citrus, grapes and stone fruits (Grieve et al., 2012). As indicated earlier, salinity adversely affects crops by a combination of mechanisms including osmotic influences, toxic ion effects (i.e. chloride, sodium and boron) and nutritional imbalances (Läuchli and Grattan, 2012). Depending upon the crop, growth stage, duration of salinity exposure and environmental conditions, some mechanisms may be more influential than others (Munns and Tester, 2008). Tree and vine crops, for example, are more susceptible to ion toxicity than most annual crops and this effect becomes more pronounced over the years and foliar injury is particularly prominent later in the season.

There is considerable uncertainty regarding the "yield-threshold" (t) soil-salinity values and that such "threshold" values, for the most part, lack physiological justification. The salinity coefficients (yield threshold (t) and slope values (s)) for the slope-threshold model of Maas-Hoffman expression (Equation 1) are determined by non-linear least-squares statistical fitting that determines the slope and threshold values from a particular set of experimental data. Despite investigators controlling salinity and minimizing all other stresses that would affect plant yield in salt tolerance studies, the standard errors associated with the 'threshold' values can be 50 to well over 100% (Grieve et al., 2012). Obviously, these large percentages represent considerable uncertainty and suggest that actual 'threshold' values do not exist (Steppuhn et al., 2005 a, b). Because of the uncertainty with the 't' value, others have suggested an ECe90 parameter (soil salinity that equates to 90% yield) as a substitute for the yield threshold parameter (van Straten et al., 2019).

Over the past few decades, scientists have since developed non-linear expressions that fit the data better and are more scientifically justified from a physiological response perspective (van Genuchten and Gupta, 1993; Steppuhn et al., 2005 a, b). The non-linear expression can be seen in Figure 1-3 and is described as follows:

$$Yr = 1 / [1 + (EC/EC_{50})^p] \quad \text{(Equation 1-2)}$$

Where, Yr is relative yield, p is an empirical shape parameter that varies between (x-y), EC is the average rootzone soil salinity expressed as the electrical conductivity of the saturated soil paste and EC_{50} is the average rootzone soil salinity where 50% yield is predicted.

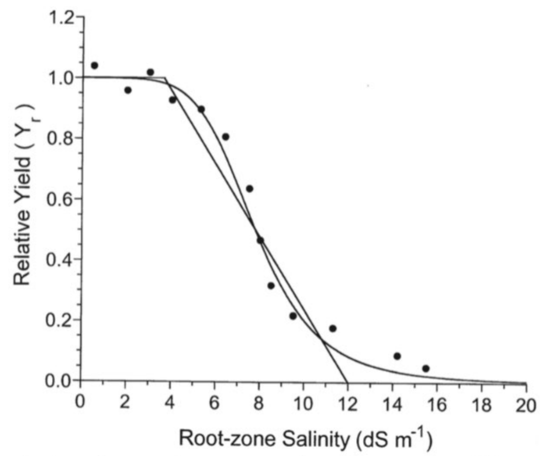

Figure 1-3. Typical Non-Linear Response Curve Superimposed on the Maas-Hoffman Slope-Threshold Model.
Source: Adapted from Steppuhn et al., 2005b.

In some cases, the response function indicates that yields of salt-tolerant crops may in fact increase slightly with mild increases in salinity and then decrease at higher levels. Despite the slightly better data fit with non-linear expressions as compared to the Maas-Hoffman 'threshold' and 'slope' model, all expressions fit the data very well. Moreover, some regulators prefer the 'threshold-slope' model since limits/guidelines can be developed using the 'threshold' value as the highest acceptable concentration that provides full crop protection.

1.2.2 Specific Ion Effects

Specific ions can affect crops that are irrigated with waste waters by several mechanisms. First, high concentrations of a given ion may cause mineral-nutrition disorders in the crop. For example, high sodium concentrations may cause deficiencies of other elements, such as potassium or calcium. Second, certain ions, such as sodium or chloride, may have toxic effects when they accumulate in tissues to lethal levels. Third, there may be specific-ion effects that promote the growth or qualitative features of the plant. This section will only focus on the first two effects.

1.2.2.1 Specific-Ion Effects: Nutritional

Salinity causes extreme ion ratios (e.g., Na+/Ca2+, Na+/K+, Cl-/NO3-) in the soil solution and thus can induce nutritional imbalances in crops. But salinity-induced nutritional disorders can vary among species and even among varieties within a species. Nutrient imbalances in the plant may result from the effect of salinity on 1) nutrient availability, 2) the uptake and/or distribution of a nutrient within the plant, and/or 3) increasing the internal plant requirement for a nutrient element resulting from physiological inactivation (Grattan and Grieve, 1999).

Nutrient uptake by crops is often reduced in saline environments but depends on the nutrient element in question and the composition and concentration of the salinizing solution (Grieve et al., 2012). The activity of a nutrient element in the soil solution decreases as salinity increases, unless the nutrient in question is part of the salinizing salts (e.g., Ca2+, Mg2+, or SO42-). For example, phosphate availability is typically reduced in saline soils by two processes: a reduction in the activity of phosphate in the solution and reduction in concentration due to sorption processes and by precipitation of Ca-P minerals. As a result, phosphate uptake and accumulation in crops is reduced in most saline environments. Regardless of salinity's effect on mineral nutrition, adding fertilizers to salt-stressed plants is not always beneficial (Grattan and Grieve, 1999).

Salinity can also cause some physiological inactivation of phosphate. Investigators found that when salt concentrations were increased, P concentration in the youngest mature tomato leaf, necessary to achieve 50% yield, almost doubled (Awad et al. 1990). In addition, they found that at any given leaf P concentration, foliar symptoms of P deficiency increased with increased NaCl salinity. This study suggests that salinity can increase the plant's internal requirement for phosphate.

Nutrient uptake and accumulation by plants is often reduced under saline conditions by competitive processes between the nutrient and a major salt species. Although plants selectively absorb K^+ over Na^+, Na^+-induced K-deficiencies can develop on crops under salinity stress by Na-salts (Janzen and Chang 1987). In addition, Cl^- salts have been found to reduce NO_3^- uptake and accumulation in crops even though this effect may not be growth-limiting (Munns and Termaat, 1986). And the opposite has been found. Nitrate can reduce Cl- uptake to the point where Cl- toxicity was reduced in citrus and avocado (Bar et al., 1997).

Economic losses of horticultural crops have been linked to inadequate calcium nutrition (Olle and Bender, 2009). Factors that affect the amount of plant-available calcium include 1) the total supply of calcium, 2) the pH of the substrate and 3) the ratio of calcium to other cations in the irrigation water (Grattan and Grieve, 1999). Calcium-related disorders may even occur in plants grown on substrates where the calcium concentration appears to be adequate. Deficiency symptoms are generally caused by differences in calcium partitioning to the growing regions of the plant. Because all plant organs (e.g. leaves, stems, flowers, fruits) actively compete for the pool of available calcium, each organ influences calcium movement independently (Läuchli and Grattan, 2012). Organs that are most actively transpiring (i.e. leaves) are more likely to have the highest calcium concentrations. Conversely, those not actively transpiring have lower calcium concentrations such as younger, developing tissue. For example, calcium deficiency appears in younger tissues showing disorders such as internal browning in heads of cabbage and lettuce and blackheart of celery (Grieve et al., 2012). Calcium deficiency disorders have also manifested in reproductive tissues, thereby reducing market quality (e.g., blossom-end rot of tomato, melon and pepper, “soft-nose” of mango and avocado, cracking and “bitter pit” of apple) (Grieve et al., 2012).

Sodium-induced calcium deficiencies have been observed in many crops within the grass family (e.g., corn, sorghum, rice, wheat and barley) where striking differences have been observed among species and cultivars. Calcium deficiency is related to some extent on sodium's effect on calcium distribution within the plant. For example, Na+ inhibits the radial movement of Ca2+ from the root epidermis to the root xylem vessels (Lynch and Läuchli 1985) and high Na+ affects Ca2+ transport to meristematic regions and developing leaves (Maas and Grieve 1987, Grieve and Maas 1988; Bernstein et al., 1993). Therefore, sodium, by some mechanism, reduces calcium's mobility in the plant.

1.2.2.2 Specific Ion Effects: Toxicity

In addition to salinity effect on mineral nutrition, specific ions i.e., Na+, Cl- and B) can cause direct injury to the crops causing further crop damage than under osmotic effects. Typically, toxic ion effects are most commonly found on woody perennials, such as tree and vine crops, while most annual, row crops remain uninjured unless salinity stress is severe. Toxic ion effects are best illustrated by Bernstein (1965) where he shows color photographs of severe leaf injury symptoms due to sodium or chloride salts in several fruit and nut crops. These crops are ineffective at excluding sodium or chloride from their leaves and because these trees live multiple years, they often suffer from toxicities at even moderate salinities (see Figure 1-4).

Figure 1-4. Salt Toxicity in Almond Leaves from an Orchard in California's San Joaquin Valley.
Source: D. Doll, UC Davis. 2015.

Chloride and sodium toxicity can damage the tree physically, biochemically and physiologically. As sodium and chloride move in the transpiration steam, they are deposited in the leaves. Older leaves have had more water transpire from them, and consequently they have higher concentrations of chloride and sodium. Once accumulated in the leaf, Na+ and Cl- typically do not remobilize to other tissues. As the concentration in the leaf increases, the salts can physically desiccate cells causing injury in the form of leaf burn. Necrotic leaves no longer photosynthesize and produce carbohydrates for the tree, which in turn, will impact growth and production. But even before salts accumulate in leaves levels that cause physical injury, the salts can reduce the chlorophyll content in leaves (Dejampour et al., 2012), and interfere with enzymatic activities affecting key metabolic pathways in both respiration and photosynthesis (Greenway and Osmond, 1972; Munns and Tester, 2008).

Although not a main 'salinizing' constituent in irrigation water, boron can also cause injury to the crop. These specific ions (i.e., Na, Cl and B) will be discussed separately below.

Sodium

Sodium can have both direct and indirect detrimental effects on plants. Direct effects are caused by the accumulation of toxic levels of Na+ in the leaves of woody species (i.e., tree crops and vines). The ability of a plant to tolerate excessive amounts of Na+ varies widely among species and rootstocks. Na+ injury on avocado, citrus, stone-fruit and some nut crops are rather widespread but can occur at Na+ concentrations as low as 5 mmol/L (115 mg/L) in soil water (Maas and Grattan, 1999), but injury may not develop until years after the trees have been exposed to brackish water. Initially, Na+ is retained in the roots and lower trunk, but after several years the Na+ entrapped in the sapwood is apparently released to the shoot after it converts to heartwood. Once the Na+ is in the transpiration stream, it can accumulate in leaves causing burn (see Figure 1-5).

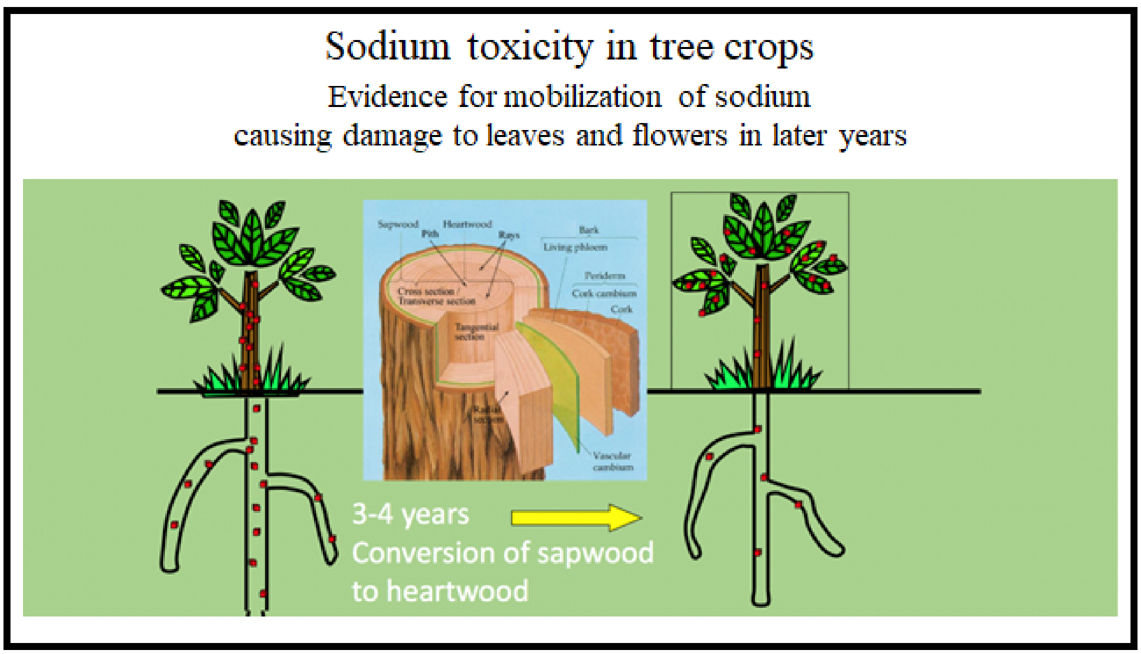

Figure 1-5. Representation of Na+ Accumulation in Sapwood and its Release to Scion after it is Converted to Heartwood.

The rootstock plays an important role in Na+ tolerance and sensitivity as well. Some rootstocks are better able to retain Na+ in the roots, trunks and branches than others allowing greater tolerance (Brown et al., 2015). In non-saline, sodic conditions where soluble Ca2+ is inadequate, Na+ toxicity would likely occur earlier.

In most annual and row crops, sodium toxicity in the form of visual injury is rarely observed. This of course implies that the soil solution has an adequate supply of soluble Ca2+. Adequate Ca2+ stabilizes root membranes allowing them to retain their integrity and selectivity (Läuchli and Epstein, 1990). Since Na+ uptake by plants is strongly regulated by Ca2+ in the soil solution, the presence of sufficient Ca2+ is essential to prevent the accumulation of Na+ to toxic levels. For these annual crops, the plants are grown and harvested before any Na+ toxicity could play a significant role, unlike perennial tree crops.

Indirect effects of Na+ include both nutritional imbalance (discussed above) and the deterioration of soil physical conditions (discussed later) (Grieve et al., 2012). The nutritional effects of Na+ are not simply related to the sodium adsorption ratio (SAR) or the exchangeable Na+ percentage (ESP) of soils, but depend upon the concentrations of Na+, Ca2+, and Mg2+ in the soil solution. In non-saline, sodic soils, total soluble salt concentrations are low and consequently, Ca2+ and/or Mg2+ concentrations can be inadequate causing poor plant growth.

Sodicity indirectly affects most crops because of the deterioration of soil aggregates affecting the overall soil structure. This topic will be described in more detail later in this review. Dispersion of aggregates affects the pore size distribution in soils thereby reducing the water infiltration rate and aeration, which negatively affect plant growth. And poorly structured soils are prone to waterlogging which promotes root disease. Therefore, crop yield reductions in sodic soils, that are not specifically sensitive to Na+, generally reflect both nutritional-imbalance problems and stresses associated with poor soil conditions.

Sodicity in soil solution or irrigation water is often assessed by estimating the SAR, which is expressed in terms of the relative concentrations of Na to that of Ca and Mg. While SAR is used widely to evaluate the sodicity hazard in many arid zones of the world, it does not capture the complexity of soil chemistry. Research and practice in recent years have demonstrated that potassium (K) and Mg, in addition to Na, can have adverse impacts on the permeability of irrigated soils (Rengasamy and Marchuk, 2011; Smith et al., 2015; Oster et al., 2016; Qadir et al., 2021).

Rengasamy and Marchuk (2011) proposed a different irrigation water quality parameter, the cation ratio of structural stability (CROSS), by including the dispersive effect of K in addition to that of Na and differentiating the flocculating effect of Mg from that of Ca (Equation 1-3).

$$CROSS = (C_{Na} + 0.56C_K) / [(C_{Ca} + 0.60C_{Mg})/2]^{0.5} \qquad \text{(Equation 1-3)}$$

Based on the water quality data of 600 water samples representing arid and semi-arid regions around the world, Qadir et al. (2021) proposed revised irrigation water quality guidelines for assessing soil permeability problems, a generalization of sodicity hazard (Figure 1-6). These guidelines are intended to cover a wide range of water quality conditions that occur in irrigated areas.

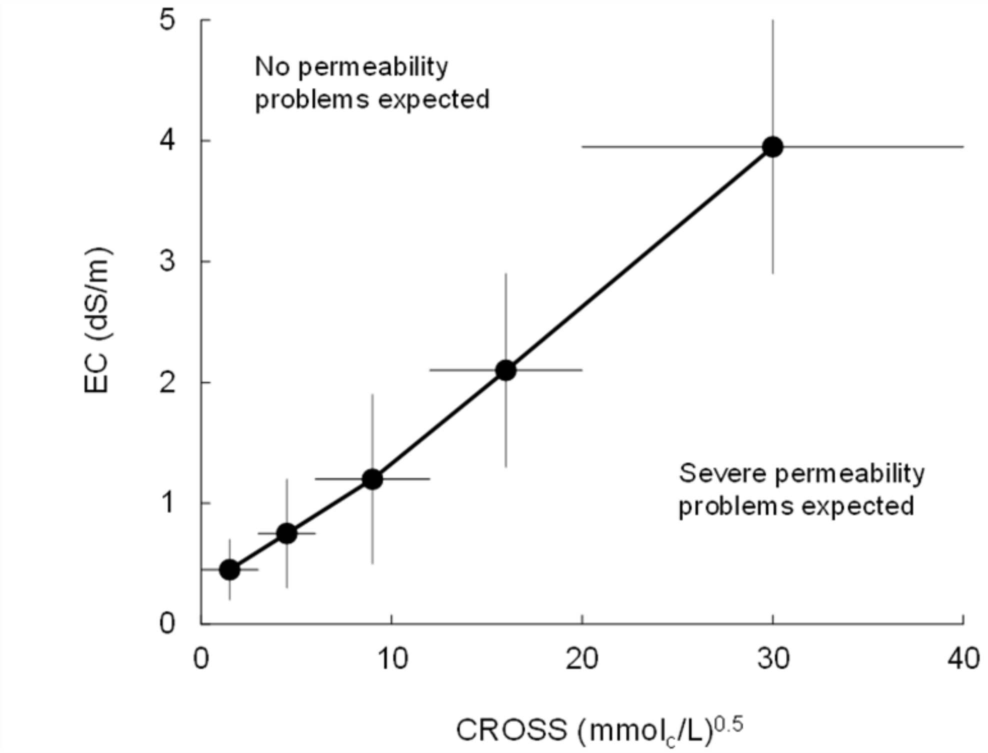

Figure 1-6. Guidelines for the Interpretation of Electrical Conductivity (EC) and the Cation Ratio of Structural Stability (CROSS) to Assess Soil Sodicity Hazard. These Guidelines Apply to Whatever Combination of A and B Coefficients are Used to Calculate CROSS. *Source:* Reprinted from *Agricultural Water Management* 255(2021); by M. Qadir, G. Sposito, C.J. Smith, and J.D. Oster; "Reassessing Irrigation Water Quality Guidelines for Sodicity Hazard"; p. 107054; Copyright (2021), with permission from Elsevier.

Chloride

Like Na^+, most annual, non-woody crops are not specifically sensitive to Cl^- even at higher concentrations (Grieve et al., 2012). However, most woody species, as well as strawberry, bean and onion, are susceptible to Cl^- toxicity, but such sensitivities are largely variety and rootstock dependent. Chloride ions move readily with the soil water, are taken up by the roots, and then move within the transpiration stream where they accumulate in leaves. And like Na^+, susceptibility to Cl^- toxicity is dependent upon the plant's ability to restrict Cl^- translocation from roots to the scion. In studies conducted over a half-century ago with avocado, grapefruit, and orange, investigators found that salt tolerance of those trees is closely related to the Cl^- accumulation properties of the rootstocks (see Grieve et al., 2012). Large differences in the salt tolerance of grape varieties have also been linked to the Cl-accumulating characteristics of different rootstocks (e.g., Bernstein et al., 1969; Ehlig, 1960). Similar effects of rootstocks on salt accumulation and tolerance have been reported for stone-fruit (Bernstein et al., 1956) and pistachio (Ferguson et al., 2002). Recent research has shown that almonds grafted on 'Nemaguard' rootstock are very sensitive to both chloride and sodium toxicity while those on 'Hansen' are considerably more tolerant (Brown et al., 2015). They found that almonds on peach-almond rootstocks were generally more tolerant than those on peach rootstocks because they restricted the uptake and translocation of these toxic ions to the scion. By selecting rootstocks that restrict Cl^- from the scions, Cl^- toxicity can be minimized or at least delayed.

The maximum Cl^- concentrations permissible in the soil water that do not cause leaf injury in selected fruit crop cultivars and rootstocks have been reported elsewhere (Grieve et al., 2012) but is included here as well (Table 1-3). While the list includes only some crops and rootstocks, it is still a valuable guide since it provides concentration ranges that are problematic to common trees and vines. Note that Cl^- sensitivity, the maximum concentration of Cl^- in the soil water above which injury occurs, covers an 8-fold concentration range (i.e. from 10 to 80 mmol/L).

Table 1-3. Chloride-Tolerance Limits of Some Fruit-Crop Rootstocks and Cultivars.
Source: Maas and Grattan 1999.

Crop	Rootstock or Cultivar	Maximum Permissible Cl^- in Soil Water without Leaf Injury† (ppm)
Rootstocks		
Avocado (*Persea americana*)	West Indian	532
	Guatemalan	425
	Mexican	355
Citrus (*Citrus* sp.)	Sunki Mandarin, Grapefruit	1773
	Cleopatra Mandarin, Rangpur Lime	1773
	Sampson Tangelo, Rough Lemon	1064
	Sour Orange, Ponkan Mandarin	1064
	Citrumelo 4475, Trifoliate Orange	709
	Cuban Shaddock, Calamondin	709
	Sweet Orange, Savage Citrange	709
	Rusk Citrange, Troyer Citrange	709
Grape (*Vitis* sp.)	Salt Creek, 1613-3	2836
	Dog Ridge	2127
Stone Fruit (*Prunus* sp.)	Marianna	1773
	Lovell, Shalil	709
	Yunnan, Nemaguard	532
Cultivars		
Berries ‡ (*Rubus* sp.)	Boysenberry	709
	Olallie Blackberry	709
	Indian Summer Raspberry	355
Grape (*Vitis* sp.)	Thompson Seedless, Perlette	1418
	Cardinal, Black Rose	709
Strawberry (*Fragaria* sp.)	Lassen	532
	Shasta	355

† For some crops, these concentrations may exceed the osmotic threshold and cause some yield reduction.
‡ Data available for one variety of each species only.

While the rootstock mainly controls the tolerance of crops to ion toxicity, research has shown that the scion (the variety grafted on the rootstock) can also have a significant role at reducing or increasing the rate of ion accumulation (Brown et al., 2015; Grattan, unpublished data, 2017).

Boron

Boron (B) is an essential micronutrient for plants but the concentration range of plant available-B in the soil solution that is optimal for growth for most crops is very narrow. Above this narrow range toxicity occurs. Criteria have been proposed to define levels that are potentially toxic and those necessary for adequate B nutrition, and yet low enough to avoid B toxicity symptoms, plant injury, and subsequent yield reduction (Ayers and Westcot 1985, Grieve et al. 2012, Gupta et al. 1985, Keren and Bingham 1985).

Boron toxicity, including how and where it is expressed in the plant, is related to the mobility of boron in the plant. Boron is thought to be immobile in most species where it accumulates within the margins and tips of the oldest leaves where injury occurs. However, boron can be re-mobilized by some species due to high concentrations of sugar alcohols (polyols) where they bind with boron and can carry it to younger tissue (Brown and Shelp, 1997). These boron-mobile plants include almonds, apples, grapes, and most stone fruits. For these crops, boron concentrations are higher in younger tissue than in older tissue and injury is expressed in the young, developing tissue such as twig die back, gum exudation and reduced bud formation. Boron immobile plants such as pistachio, tomato, walnut, and fig, do not have high concentrations of polyols and the boron concentrates in the margins of older leaf tissue (see Figure 1-7). Injury in these crops is expressed as the classical necrosis on leaf tips and margins.

Figure 1-7. Boron Injury on the Margins of 'Kerman' Pistachio Leaves in an Orchard in California's San Joaquin Valley (B-immobile Species).
Source: Photo by S.R. Grattan, UC Davis.

Much of the guidelines that were developed that identify boron sufficient and excessive ranges for crops are based on data from experiments conducted during 1930-34 by Eaton (1944). While useful, these experimental data cannot be used to develop any reliable growth response function with increasing solution boron. Nevertheless, his results provide the majority of the threshold limits, above which injury occurs, presented in Appendix A.2. In several cases, plant response to excess B was fitted to the two piece linear response model that was used for crop salt tolerance (see Grieve et al., 2012). Therefore, the table in Appendix A.2 does provide the threshold and slope parameters for these limited crops where the 'threshold' is the maximum concentration in the soil water before yields are reduced just as described for crop salt tolerance. Like salt tolerance, B tolerance varies with climate, soil conditions, and crop cultivars; therefore, the data presented in Appendix A.2 may not apply to all cultural conditions.

1.3 Future Research Directions

The linear and non-linear models that describe relative crop yield response to rootzone salinity used data primarily from field-plot studies. Crops in most of these studies were irrigated frequently under furrow or flood irrigation, and used high leaching fractions to avoid crop water stress. This is intentionally done in field studies to create a salinity profile that changes little over space and time. With uniform conditions, it is easier to compare tolerances among crop species than it would under conditions with salinity varies over space and time.

While creating uniform, steady state rootzones provides an opportunity to compare crops for different tolerances to salinity and rank their sensitivity, such uniform profiles are uncharacteristic of field soils (Homaee and Schmidhalter, 2008). Field soils develop characteristic salt distribution patterns (Figure 1-8). These patterns are a result of water movement via gravitational and capillary action and subsequent root water extraction and soil evaporation. Under sprinkler or border irrigation, the salinity increases with depth while under furrow or drip, salinity increases horizontally in the direction of water flow in addition to increases in the vertical direction.

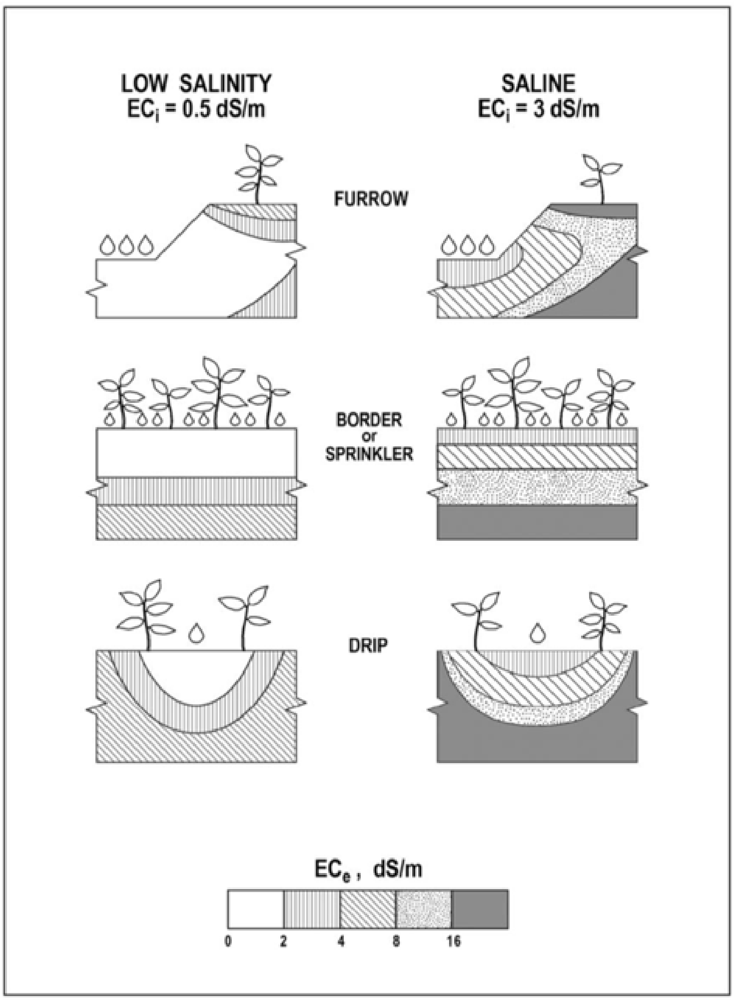

Figure 1-8. Characteristic Salt Distribution Patterns in Soils in Fields with Different Irrigation Systems.
Source: Hoffman et al., 1990.

The current salt tolerance data are based on crop response to the saturated soil extract (ECe) when in reality, the crop is responding to the salinity in the soil water (ECsw) which is continuously changing over space and time. In reference to Figure 7, which illustrates spatial salinity distribution as influenced by the irrigation method, the soil water content changes over space and time. Over the past several decades as water scarcity continues in impact crop production, shifts from conventional, surface irrigation methods to more efficient low-pressure systems (i.e., drip and mini-sprinklers). Studies have shown that crops under drip irrigation are more tolerant to salinity than they are under conventional methods (Bernstein and Francois, 1973; Hillel, 2005). Under high frequency drip irrigation, the salinity of the soil water near the dipper is close to that of the irrigation water and the water content is often above field capacity, particularly for a short period after irrigation. Therefore, the roots are responding to a lower soil salinity than they would under conventional irrigation practices. While the soil volume is less, high frequency irrigation can allow the crop to maintain its crop water needs. This brings into question the validity of applying the salinity coefficients (slope, s and threshold, t), developed under conventional systems and expressed on an average rootzone ECe bases (see Appendix A.1), to high frequency drip irrigation practices. This dynamic condition complicates how best to characterize the rootzone as the roots are exposed to changes in soil water content and salinity in different parts of the profile.

It has been recognized for decades that the major root activity is found in the least saline portions of the soil profile (USDA-USSL Staff, 1954). Water uptake by roots in the least saline portion of the rootzone is key to shoot growth where shoot biomass can be 3-10 fold higher in heterogeneous soil profiles than under equivalent homogeneous salinity conditions (Bazihizina et al., 2012). But root length density and root water extraction play an important role. Experiments with alfalfa have shown that while the water uptake rate by roots reacts to soil salinity, 'root activity' and 'evaporative demand' become more important factors in controlling the uptake pattern (Homaee and Schmidhalter, 2008).

Roots will grow and develop in the most favorable portions of the rootzone considering factors such as salinity, water content, nutrients, pH, oxygen availability, soil strength, disease pressure, etc. Plant roots indeed exhibit a remarkable plasticity in their developmental response to variable soil conditions (Rewald et al., 2013). But the degree of plasticity in likely related to genetic factors as well as stressor extremes in the rootzone. Research is lacking in this general area of root plasticity to variable soil conditions.

The distribution of abiotic and biotic stressors will also vary throughout the profile. For example, soil salinity may be low in the upper portion of the soil profile but soil water content (i.e., matric potential) will vary widely due to higher root length density. In the lower portion of the soil profile the salinity can be substantially higher (i.e., low osmotic potential) but water content is higher and fluctuates less due to lower root activity. When multiple stresses occur simultaneously, it is the dominant stress that largely controls crop growth and response (Maas and Grattan, 1999; Shani et al., 2007). Likewise, the release of the most dominant stress will promote the most growth. Nevertheless, there is considerable uncertainty how the plant integrates multiple stresses over space and time. More research is needed to better understand the physiological mechanisms underlying plant water relations and shoot ion regulation in plants under heterogenous salinities (Bazihizina et al., 2012) and how roots are able to adapt over the season with changing conditions. While there will likely be complex interactions, it is nonetheless an important area of future research. As new varieties and rootstocks continue to be developed, more research is needed to assess the tolerance to specific ion accumulation and tolerance. Currently, most of the data on tolerances to trees, vines and rootstocks are decades old.

Research at the cellular and genetic level will continue in the future. Quantitative trait locus (QTL) mapping analysis from a cross between parents of contrasting tolerance is the method most promising for development of more salt tolerant cultivars (Mujeeb-Kazi et al., 2019). However, for salt tolerance in multi-genetic (Flowers and Flowers, 2005; Munns, 2005) there has been only minor progress in developing cultivars that are more successful in the field. Finally, is the need for research on the impact of direct water reuse on the ionic composition of irrigation water and its potential impact on soil and plants. Direct water reuse often implies water from RO membrane or AOP (Advanced Oxidation Process). The resulting reused water might arrive at the irrigation field through two different mechanisms (a) domestic water use followed by advanced wastewater treatment, or (b) direct transport to the irrigation field. Direct water reuse is under quick development in various parts of the world due to the growing need for water and needs additional scientific attention.

1.4 Summary

Salinity has impacted irrigated agriculture and civilizations for millennia. The US Salinity Laboratory was instrumental in studying salinity effects on soil and plant systems in the first half of the 20th century and setting the basic foundation of our general understanding. However, researchers across the globe have since made substantial advances in how salinity and salt tolerance is characterized, how the plant is affected by salinity and specific ions, and the role of the cells and membranes in excluding toxic ions and adjusting osmotically to the saline soil solution. And some, yet limited, breeding successes have developed cultivars that are more tolerant to salinity. Nevertheless, more research is needed to develop cultivars that are more tolerant in field conditions and to better characterize plant response in heterogeneous soil conditions, particularly as drip and other low-pressure irrigation systems become more and more prevalent. This new information is critical as recycled water produced by various technologies continues to expand in arid and semi-arid climates.

CHAPTER 2

Management of Saline-Sodic Soils under Recycled Water Irrigation

2.1 Introduction

Recycled water needs to be managed to provide the optimal use of this resource and to ensure that crop yield is maximized while minimizing crop stress, energy use, and losses of nutrients to surface and groundwater sources. These Good Agricultural Practices (GAPs) will vary depending on the irrigation water quality, the amount applied, the crop and soil type, the irrigation method used, and site-specific conditions. In Chapter 2, the focus will be on irrigation methods and management, irrigation quality and crop salt tolerance, leaching and drainage, and managing sodicity to sustain soil physical conditions. While fertilizer and pest management practices are indeed important GAP considerations, they are beyond the scope of this chapter and will not be addressed here.

Irrigation scheduling, regardless of whether the water is recycled or not, is critical to assure that the right amount of water is applied to the crop, as uniformly as possible, and at the correct time. Throughout the season, the irrigation supply should replenish water losses from the root zone via evapotranspiration (ET) and drainage avoiding the depletion of soil water below the critical limit. When using brackish water, it is particularly critical that soil moisture remains at a higher matric potential (less dry) than would be tolerated using non-saline water and that the concentration of salts in the soil water is maintained within tolerable levels.

There are several methods for scheduling irrigation, and many are not mutually exclusive. Some methods monitor the plant and soil such as those based on the monitoring soil salinity and moisture content (e.g., gravimetric soil moisture sampling, dielectric sensors, and soil salinity probes, measuring soil moisture tension (e.g., tensiometers, electrical resistance blocks, etc.), and characterizing plant response to soil water status (e.g. monitoring stem-water potential, canopy temperature, sap flow, and plant growth rate). Most of these are useful for the timing of irrigation. Other methods rely on weather data, canopy cover, and irrigation management practices to estimate ET. It is the water balance approach to irrigation scheduling that is useful in determining the amount of water to apply as it requires the use of weather parameters and formula (e.g., Penman-Monteith (equation) to quantify crop evapotranspiration (ETc) using reference ETo and site-specific crop coefficients (Kc) (Allen et al., 1998). More recently ETc from remote sensing is beginning to be provided as a commercial service to growers.

2.2 Past and Current Knowledge

2.2.1 Importance of Leaching for Salinity Control

Soil salinity is controlled by avoiding excessive salt accumulation in the crop root zone. The sustained, long-term use of saline water for irrigation, therefore, requires salt to move past the root zone. This downward movement is commonly referred to as leaching and is necessary regardless of plant type to optimize plant productivity. The leaching fraction (LF) is defined as the fraction of infiltrated irrigation water that drains below the root zone. Simply, it is the volume of drainage water divided by the volume of infiltrated water.

$$Leaching\ Fraction\ (LF) = \frac{volume\ of\ drainage\ water}{volume\ of\ infiltrated\ water} \qquad \text{(Equation 2-1)}$$

The LF needed is dependent on plant tolerance to salinity, the salinity of the irrigation water, crop evapotranspiration, and site-specific conditions. The leaching requirement (LR), on the other hand, is the minimum LF needed to maintain the soil salinity at the 'threshold' ECe level (t) for the crop type being irrigated (Ayers and Westcot, 1985). The greater salt tolerance, the lower the required leaching; and for a given salt tolerance, the higher the irrigation water salinity, the greater the required leaching.

When leaching occurs, soil salinity increases with increased depth in the soil profile as shown in Figure 2-1. But the increase in salinity with depth is dependent upon the irrigation water salinity, the LF, and the root water extraction pattern. Figure 2-1 shows two distinct soil salinity profiles in an alfalfa field; one using a saline water of 6 dS/m and a high LF of 50% and the other a lower salinity water of 2 dS/m and a lower LF of 7% (Hanson et al., 2006). Note that the average root zone salinity in this alfalfa field under both scenarios are, more or less, equivalent to one another despite the fact that one irrigation water is three times the salinity of the other.

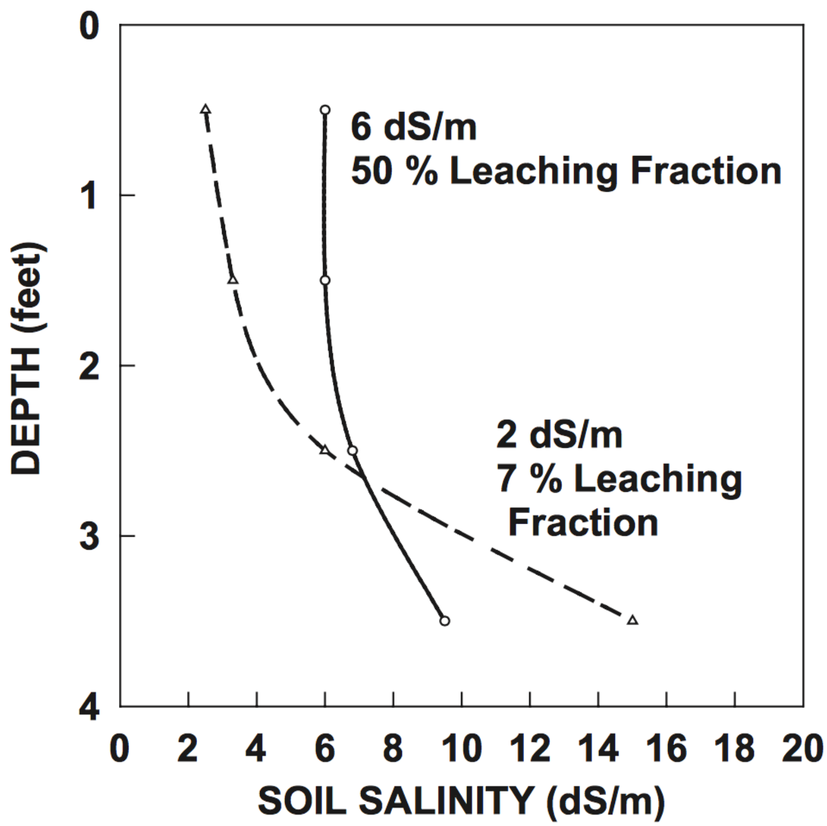

Figure 2-1. Salt Distribution in an Alfalfa Field Irrigated with Different Water Salinities and Leaching Fractions.
Source: Hanson et al., 2006. Note: 1.0 ft = 30.5 cm.

2.2.1.1 ECw-ECe-LF Relations under Conventional Irrigation

The difficulty with LF is measuring the volume of drainage water under field conditions. But this difficulty can be overcome by developing relationships between ECw, LF and average root zone salinity (ECe). In order to effectively use the salt tolerance tables (A 2-1 and A 2-2) presented in Appendix A, a relationship of this type is needed. Relationships between ECw (electrical conductivity in the irrigation water) and ECe (average root zone salinity expressed as the EC of the saturated soil extract) were developed by Ayers and Westcot (1985). They assumed crops are irrigated by conventional methods (i.e. irrigations are infrequent where 50% or more of the available water is depleted between irrigations) and a steady-state LF is achieved (Figure 2-2). Steady-state leaching assumes that the flux of water downward in the soil profile is constant and that the leaching fraction remains fixed. Figure 2-2 was constructed based on the infinite number of scenarios from relationships illustrated in Figure 2-1. Note that as the LF increases, the slope of this relationship decreases. Ayers and Westcot (1985) also assumed that the root water extraction pattern would follow a 40-30-20-10 relationship indicating water uptake for the top, second, third and bottom quarters of the root zone are assumed to be 40, 30, 20 and 10%, respectively.

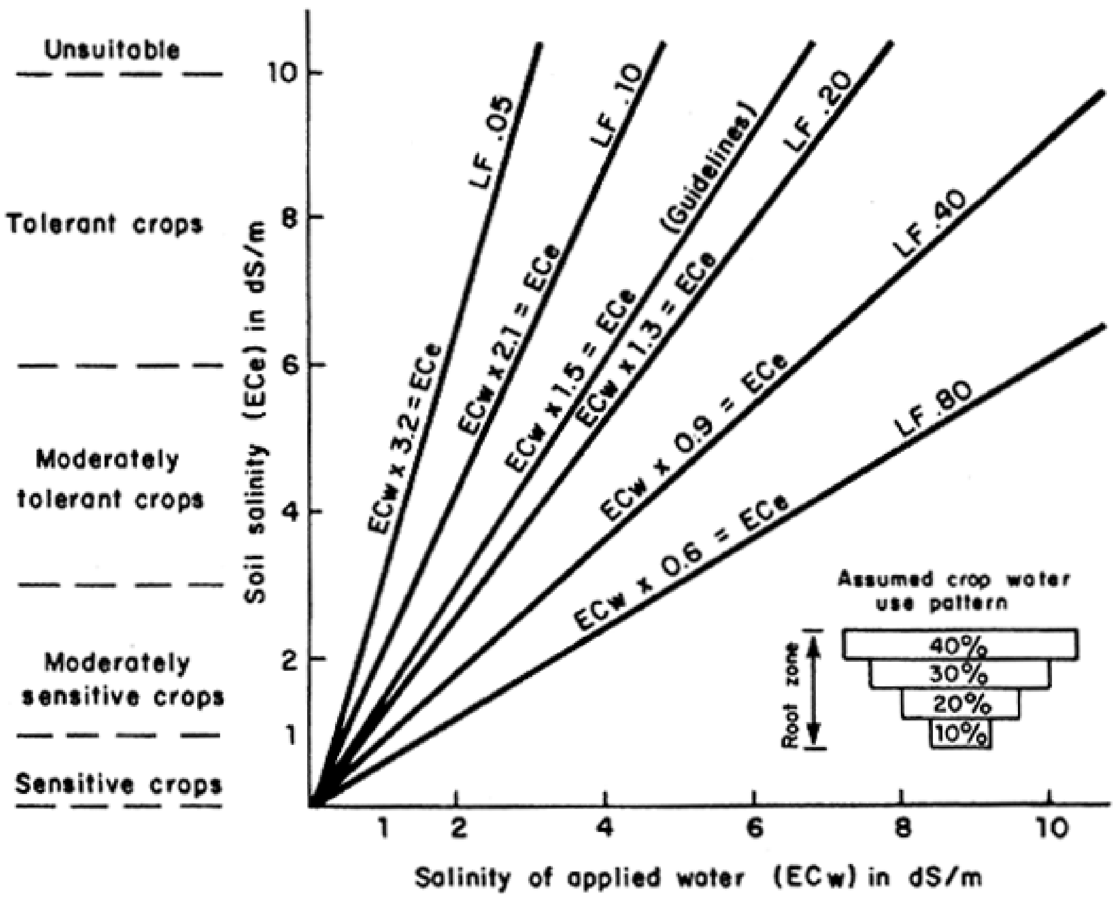

Figure 2-2. Relationship between Soil Salinity (ECe) and Salinity of the Applied Irrigation Water (ECw) under a Series of Steady-State Leaching Fractions (0.05 to 0.80).

Source: Adapted from Ayers and Westcot 1985 with permission from FAO.

Rather than trying to interpret ECe values based on ECw and assumed steady-state LF off the graph, a table with different concentration factors (Fc) for different LFs was developed by Suarez (2012) (Table 2-1). This relationship applies to conventional irrigation practices. This Fc is basically the slope of the relationships in Figure 2-2 such that ECe = (Fc) ECw.

Table 2-1. The Concentration Factor (FC) in Relation to the Leaching Fraction (FC) Assuming a 40%-30%-20%-10% Root Water Extraction Pattern with Descending Root Zone Quarters and Assuming a Linear Average (Suarez, 2012). To Be Used for Lower Frequency, Conventional Irrigation Such as Surface.

Leaching Fraction (LF)	Concentration Factor (Fc)
0.05	2.79
0.10	1.88
0.20	1.29
0.30	1.03
0.40	0.87
0.50	0.77

To better illustrate how this relationship can be applied to crops with different sensitivities to salinity, they placed general 'salt tolerance' categories on the y-axis to indicate the soil salinity threshold (t) limits where yields begin to decline. For example, if an irrigation with an ECw of 4.0 dS/m is used with an achievable LF of 40%, then the expected average root zone salinity (ECe) would be 3.5 dS/m (see Figure 2-2 and Table 2-1). This suggests that only crops classified as 'moderately tolerant' or 'tolerant' to salinity can be grown with this water and LF without a reduction in the yield potential. Using the crop salinity threshold 't' from Appendix A.1 and A.2, the ECw can be calculated indicating the maximal salinity the irrigation water can be to achieve the full yield potential of a crop, given this leaching-fraction (LF). For example, if the yield threshold ECe is 2.5 dS/m, as it is for tomato, then the maximum ECw that can be used to achieve full-yield potential assuming a 10% LF is 1.3 dS/m. Irrigation waters of a higher salinity can be used to irrigate tomato, but the full potential may not be achieved or a higher LF is needed.

2.2.1.2 ECw-ECe-LF Relations for High-Frequency Irrigation

Recycled water is commonly used for high frequency irrigation of horticular crops. Similar ECw-ECe-LF relationships have also been developed with high-frequency irrigation, such as drip irrigation (Figure 2-3). Using this figure, if a leaching fraction of 10% could be maintained using drip irrigation with an irrigation water with an ECw of 3.0 dS/m, the average rootzone salinity (ECe) would be 4 dS/m. Under conventional irrigation, this same water and leaching fraction would produce an ECe of 6.3 dS/m (see Figure 2-2). The difference between the high frequency and conventional methods is that the average root zone soil salinity is calculated differently. For conventional irrigation, the average root zone salinity is the simple average of the ECe in the 1st, 2nd, 3rd and bottom quarters of the root zone. For high frequency irrigation, the average root zone salinity for the 4 root zone quarters are weighted based on water uptake; where the water uptake for the top, second, third and bottom quarters are assumed to be 40, 30, 20 and 10%, respectively (Hanson et al., 2006). Therefore, salinity in the upper quarter has 4 times the weight as that in the bottom quarter. Similar to the previous table, Table 2-2 presents Fc values for different LFs based on a root-water-uptake-weighed root-zone salinity.

Table 2-2. The Concentration Factor (FC) in Relation to the Leaching Fraction (FC) and Percentage of Applied Water Assuming a 40%-30%-20%-10% Root Water Extraction Pattern with Descending Root Zone Quarters and Assuming a Water Uptake Weighted Root Zone Salinity (Rhoades et al., 1992; Suarez, 2012). To Be Used for High Frequency Irrigation Such as Drip.

Leaching Fraction (LF)	Concentration Factor (Fc)
0.05	1.79
0.10	1.35
0.20	1.03
0.30	0.87
0.40	0.77
0.50	0.70

The relationships in Figure 2-3 would also be different if the uptake function were changed. If the root water uptake followed an exponential pattern (i.e. 71-20-6-2), such as that described by Skaggs et al. (2014), the slopes of each of the lines would be even less than indicated here implying that waters of even higher salinity can be used. That is, the average root zone salinity would be less because the upper quarters of the profile, where salinity is less, are weighted more. High frequency drip and micro-sprinkler irrigation will allow poorer quality waters to be used than could be used by other irrigation methods. Caution is advised because reclamation leaching may be needed at some point to leach salts, or boron, from the root zone during winter months (see section on Reclamation Leaching).

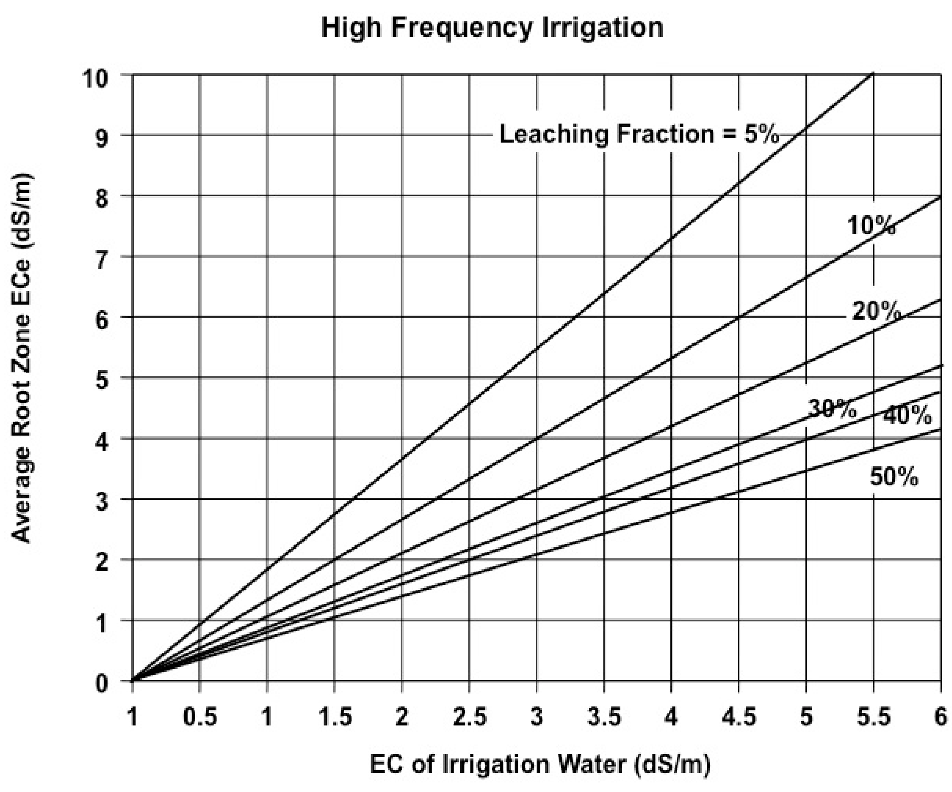

Figure 2-3. Relationship between EC of the Irrigation Water and the Average ECe of the Rootzone under High Frequency Irrigation (i.e., Drip and Micro-sprinklers). *Source:* Hanson et al., 2006.

2.2.2 Irrigation Water Boron Versus Soil Solution Boron

Unlike salts, boron has a high affinity for the soil and therefore the relationship between boron in the soil solution vs that in the irrigation is not as straight forward. If boron in the recycled water is at concentrations that could be potentially damaging to the crop, the boron in the soil solution will be controlled by sorption processes by the solid phase that keep the concentration below toxic levels. However, once all adsorption sites are saturated with boron, boron in the soil solution will begin to behave like typical salts. This may take a number of years to reach adsorption saturation. As such, plants may tolerate higher boron concentrations in the irrigation water at early times before B-adsorption is saturated.

This complexity between boron in the irrigation water vs that in the soil solution was known to Ayers and Westcot (1985). Because plant tolerance to boron is based on the concentration of boron in the soil solution (Appendix A 2-3), they approximated that the concentration of boron in the soil solution was about equivalent to that of the irrigation water, or slightly higher. A group of Canadian researchers developed an approach to relate these two concentrations (Jame et al., 1982; Leyshon and Jame, 1993). Using the principals of mass balance, they developed an approach to estimate the B concentration of the soil water based on B concentration in the irrigation water for a given leaching fraction. They assumed that B uptake was directly proportional to root distribution where they assumed the root water uptake followed the 40-30-20-10 pattern in descending root zone quarters. They found that the rootzone weighted average of B in the soil solution (Bss) was 1.4-1.9 times that of the irrigation water containing 0.5-10.0 mg/L B if the LF was 25%. But if the LF was only 10%, Bss would be 1.9-2.7 times higher than that of the irrigation water. And there was considerable time to reach equilibrium, which is dependent upon soil texture. For example, irrigation water containing 1.0 mg/L B, it would take between 10 (sandy loam) to 55 years (clay loam) if the LF was 25% and the initial soil B concentration was zero.

The relationship between B concentration in the irrigation water to that in the soil solution is illustrated in Figure 2-4. These linear curves were developed from the modeling data reported by Jame et al., 1982. As the LF increases, the same B concentration in the irrigation water produces a lower Bss concentration.

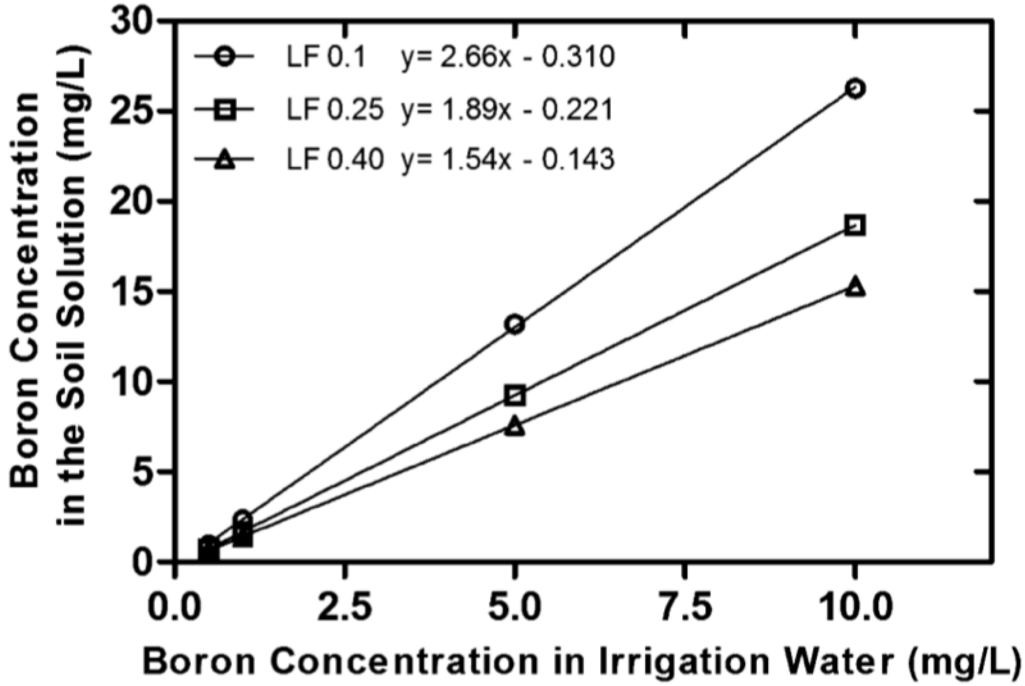

Figure 2-4. The Relationship between the Concentration of B in the Irrigation Water (Bw) and the Root-Zone-Weighted Concentration in the Soil Solution (Bss).

Source: "Boron Concentration of Irrigation Water (mg/L)" by Jame et al., is licensed under a Creative Commons license.

2.2.3 Limitations to the Leaching Fraction Concept

The leaching requirement is an attractive concept but has serious limitations. First, the leaching fraction expression has no time element. Therefore, there is no accounting for how long leaching will take, which will differ depending on the permeability of the soils. Second, the evapotranspiration (ET) of the crop is assumed to be independent of the average root zone salinity. As a result, calculated crop water requirements will be over-estimated when the average root zone salinity exceeds the threshold salinity of the crop, which corresponds to a yield potential of less than 100 percent (Letey and Dinar, 1986; Shani et al., 2005, Letey and Feng, 2007). That is, a salt-stressed crop will use less water than a non-stressed crop. Consequently, crop ET will be reduced, and leaching, with the same quantity of applied water, will be increased. Other issues also affect the proper calculation of crop water requirements: 1) initial levels of salinity in the root zone, 2) spatial variation in the amount of water applied, 3) the amount that infiltrates into the soil and 4) the difficulty of achieving adequate infiltration in a field to achieve the desired leaching fraction. And in drip irrigated fields, actual LFs are difficult to quantify because LF, soil salinity, soil water content and root density all vary with distance and depth from the drip lines (Hanson et al., 2008). Nevertheless, leaching does occur in drip irrigated fields, but the zone of leaching is directly below the emitter.

In light of the discussion above, recent studies have shown that the ECw - ECe relations described by Ayers and Westcot (1985), which are based on steady-state LF conditions, tend to be too conservative and overestimate soil salinity and therefore overestimate yield losses in most cases (Corwin et al., 2007; Corwin and Grattan, 2018; Letey et al., 2011). These scientists suggest that transient-state models have the potential to more accurately predict soil salinity, as well as soil Cl^-, Na^+ and B. There are many models that predict soil water changes in the root zone and crop response, but all vary in function and complexity. Such models include ENVIRO-GRO (Feng et al., 2003), HYDRUS (Simunek et al., 2008), TETrans (Corwin et al., 1990), SALTMED (Ragab et al., 2005 a,b), SWAP (van Dam et al., 2008), UNSATCHEM (Suarez and Simunek, 1997) among others. However, these transient models are complex and most require detailed site-specific information. And there are uncertainties regarding how the crop responds to salinity and soil water content that varies in the root zone over space and time. Therefore, the steady-state leaching approach remains a valid approach that can be used with recycled water and is a conservative estimate of the leaching requirements.

Despite these limitations of the leaching fraction concept, in order to control salinity, leaching must occur whether it is achieved at the beginning, during the season, or at the end of the crop season (Ayers and Westcot, 1985; Shalhevet, 1994). To allow this, soil physical conditions must be maintained such that adequate water to satisfy by crop ET must readily enter the soil. This is primarily a problem when the water used for irrigation is sodic or saline-sodic, where low infiltration rates into the soil restrict the water necessary to meet the crop water requirements and extra water necessary for leaching.

2.2.3.1 Improving Soil Physical Properties

Soil physical properties can be altered by irrigation with saline-sodic recycled water, made apparent when good quality water is used or rainfall occurs after saline-sodic water application (Oster and Jayawardane, 1998; Oster et al., 1999; Shainberg and Letey, 1984). Potential adverse effects include reduced infiltration and redistribution within the soil, poor soil tilth, and inadequate aeration resulting in anoxic conditions for roots (Oster et al., 1999). These negative impacts, however, can be reduced with appropriate soil and water amendments like gypsum, sulfur, and sulfuric acid (Oster et al., 1992). The goal in any amendment is to maximize the free Ca^{2+} in the soil solution. Therefore, a direct calcium supplier (e.g., gypsum) or an acidifying amendment (such as elemental sulfur, sulfuric acid, urea sulfuric acid (NpHuric) or lime sulfur) to dissolve calcite ($CaCO_3$) in the soil to form free Ca^{2+} are recommended (Hopmans et al., 2021). In addition, if high levels of B are present in the water, its accumulation in the soil could adversely affect crop production (Grattan and Oster, 2003). The need to leach salts and B from the root zone will also leach NO_3^-. Nitrate losses can be mitigated by additional fertilizer application, but such losses are non-economical and could be environmentally damaging. If, however, leaching can be done at the end of the season when salinity is maximal and soil nitrate concentration is minimal, this would reduce the environmental impact of nitrate contamination of groundwater while at the same time controlling salinity.

Incorporation of organic matter to the soil can also affect soil physical conditions. Taylor and Olsson (1987) and Quirk (1978) demonstrated that increased levels of organic matter arising from pasture root systems stabilize soil structure after gypsum is no longer present at the soil surface in sufficient amounts. The adoption of farming practices such as minimum tillage leads to increased retention of crop residues in the form of surface mulches. This encourages soil microbial activity including the production of exopolysaccharides (EPS) that increase and maintain the continuity of large biopores which are effective at conducting water and air to subsoils (Jayawardane and Chan, 1994).

2.2.4 Drainage Systems

The role of drainage systems in the management of saline soils is particularly important especially when salinity problems are associated with the presence of shallow water table or impermeable soil layer close to the surface causing waterlogging conditions. The presence of a shallow water table may directly influence the soil water balance and the presence of salts in the root zone through the upward capillary flow of water from the saturated into the unsaturated zone. In such conditions, a salt balance cannot be achieved in the root zone. A subsurface drainage system consists of laterals (often referred to as 'tile' lines) that consist of corrugated plastic tubing with perforations allowing saturated water to flow into the line. The laterals are buried throughout the field at a specified depth and spacing and are connected to a mainline.

Well-designed drainage systems allow the downward movement of water through soils and lower the water table to a desirable level. The goal is to lower the saline water table to a depth so it does not contribute to the transport of salts into the root zone by capillary rise. In such a way, controlling the groundwater table, the drainage system provides adequate aeration of the root zone and improves the soil conditions for growing plants. Installing drainage laterals too deep is undesirable in that more drainage water would need to be managed. There are many drainage engineers that have formulas for designing drainage systems. For more information on improving subsurface drainage systems, understanding water table depth criteria for drain design, interceptor drains and designing relief drainage systems see Hanson et al. (2006).

2.2.5 Reclamation Leaching

Researchers have observed that leaching, in many cases, is more effective at the end of the season rather than trying to impose a LF for each irrigation, especially in fields with low permeability. In many soils, the infiltration rate diminishes throughout the season and the best opportunity to leach the soil is after the growing season when the evaporative demand is low. Several decades ago, Hoffman (1986) proposed that sprinkler irrigation or intermittent ponding was the most effective means at leaching salts from the soil and developed a leaching reclamation curve (Figure 2-5). This reclamation leaching approach by these methods was found to be independent of soil type. The reclamation curve plots the depth of leaching water needed (d_l) per depth of soil (d_s) vs the ratio of the desired soil salinity (C) to the initial soil salinity (C_0).

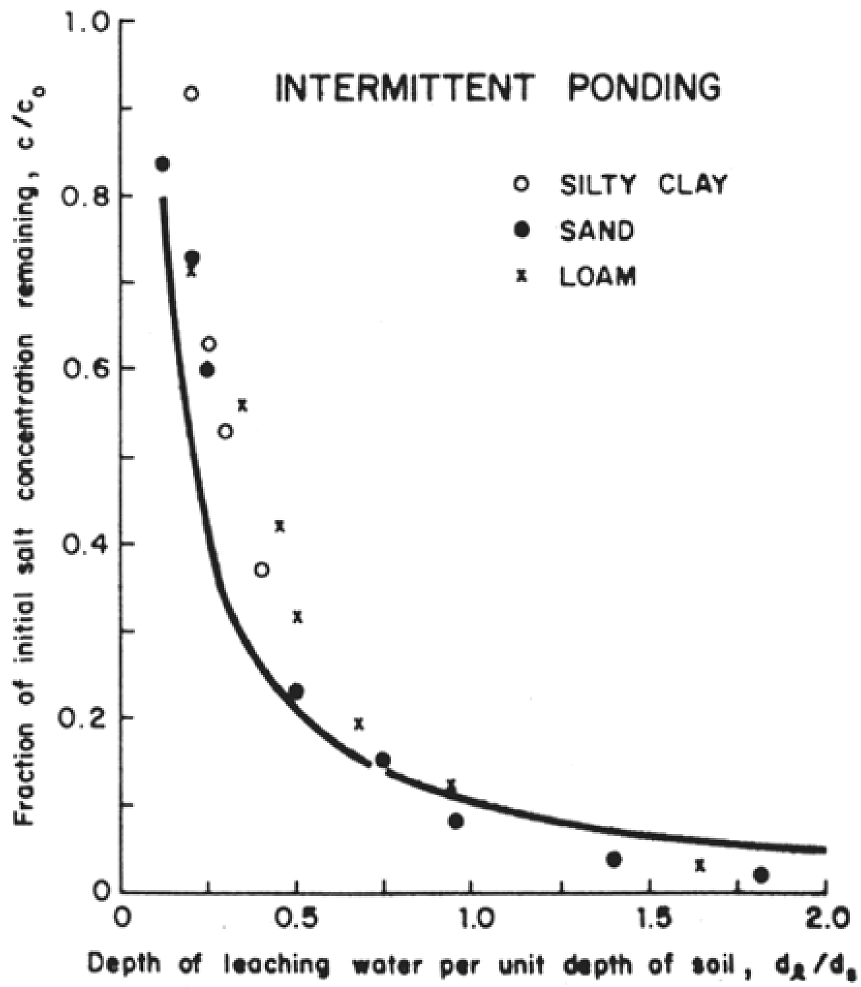

Figure 2-5. Reclamation Leaching Function under Sprinkler Irrigation or Intermittent Ponding.
Source: Ayers and Westcot, 1985 with permission from FAO.

This reclamation curve can be used in the following way. Suppose the average root zone salinity (ECe) in the top meter of soil was 6.0 dS/m and the target is to reduce the soil salinity in this 100 cm profile to 3.0 dS/m. Therefore, the fraction of salt reduction is 0.5 (3.0 dS/m/6.0 dS/m). According to the graph, the amount of leaching water needed is 0.25 meters of water for every meter of soil. Therefore, 25 cm of water would have to be leached by either sprinkler irrigation or intermittent ponding to reduce the soil salinity in the top meter to 3.0 dS/m. While this is a valuable tool, it does not replace the importance of soil samples before and after the reclamation process to determine how close the final soil salinity is to the targeted soil salinity.

It is recommended that this reclamation leaching practice also consider rainfall. Rainfall, depending upon the location where recycled water is used for irrigation, can be significant at reducing soil salinity. In many Mediterranean climates where rainfall is more substantial in certain months of the year, reclamation leaching would be more effective at occurring after the rain period.

Reclamation of saline-sodic soils requires an additional step. Before reclamation can be effective, the sodicity of the soil must be reduced to improve soil structure. Only an improvement in soil structure will allow the pore size distribution to be adequate to promote drainage and thus adequate leaching. A more detailed discussion on amendments to reduce soil sodicity can be found in Hanson et al. (2006), Ayers and Westcot (1985) and Hopmans et al. (2021).

A reclamation curve is also presented in Ayers and Westcot (1985) for soil boron but typically it takes several times the amount of water to reduce soil boron by the same percentage in comparison to soil salinity. This is due to boron's affinity for the soil surface. A much more detailed discussion on the reclamation of saline, sodic and boron-affected soils can be found in a chapter by Keren and Miyamoto (2012).

2.2.6 Irrigation Methods

The method of irrigation can have a profound influence on how salt is distributed in the soil profile and how the crop responds to the applied irrigation water. The terrain can dictate to some extent what system can be used. Surface methods are limited to flat, level landscapes but undulating landscapes require pressurized systems such as sprinkler and drip. Well-designed sprinkler and drip systems typically have higher achievable distribution uniformities (DUs) than do surface methods. With higher DUs, not only is irrigation water spread more uniformly over the surface but water is used more efficiently as less water is lost to deep percolation losses. The most suitable irrigation system should be used according to site- specific conditions as there is no one irrigation system that can fit all. For more information on irrigation efficiency and optimizing DUs, refer to Hanson et al (2004).

2.2.6.1 Salt Distribution under Different Irrigation Methods

The salt distribution patterns vary considerably among methods of irrigation. The pattern of salt distribution, influenced by the different irrigation methods, affects where the roots proliferate in the soil profile. Figures 2-6 and 2-7 show typical soil salinity distribution patterns under different irrigation methods that were previously mentioned in Section 2.2.1.

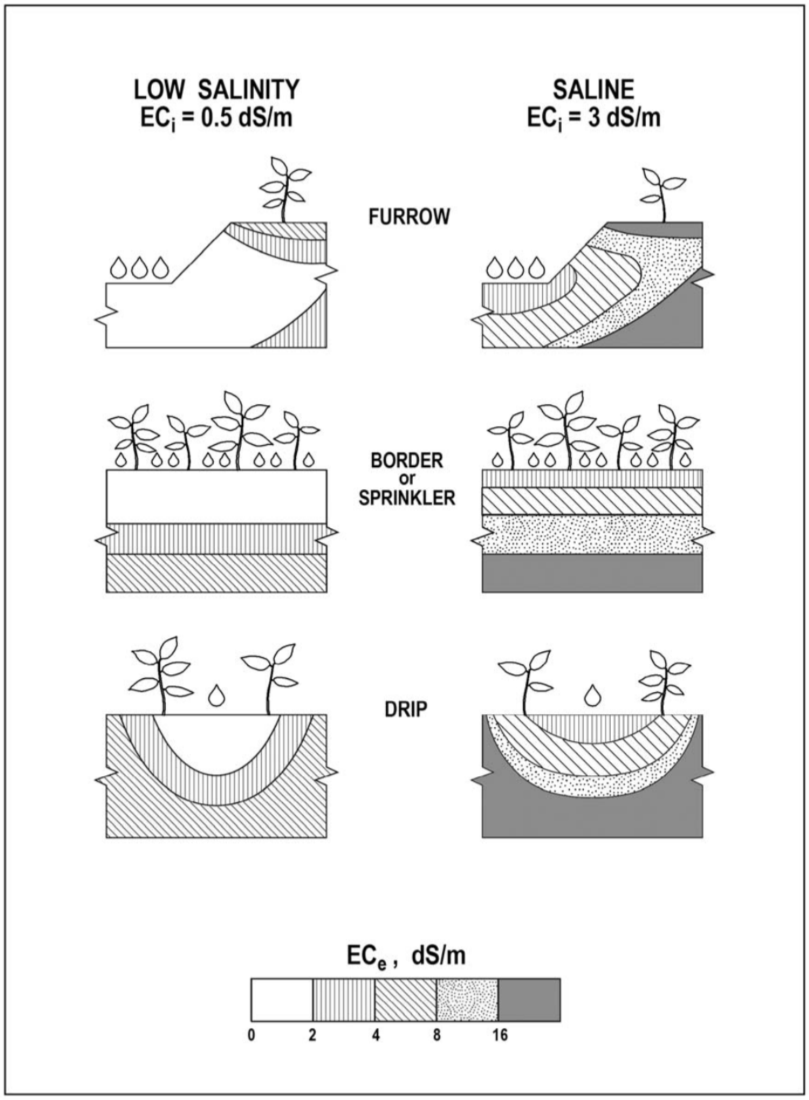

Figure 2-6. Characteristic Salt Distribution Patterns in Furrow Irrigated, Border or Sprinkler, and Surface Drip Irrigated Fields.

Source: Hoffman et al., 1990.

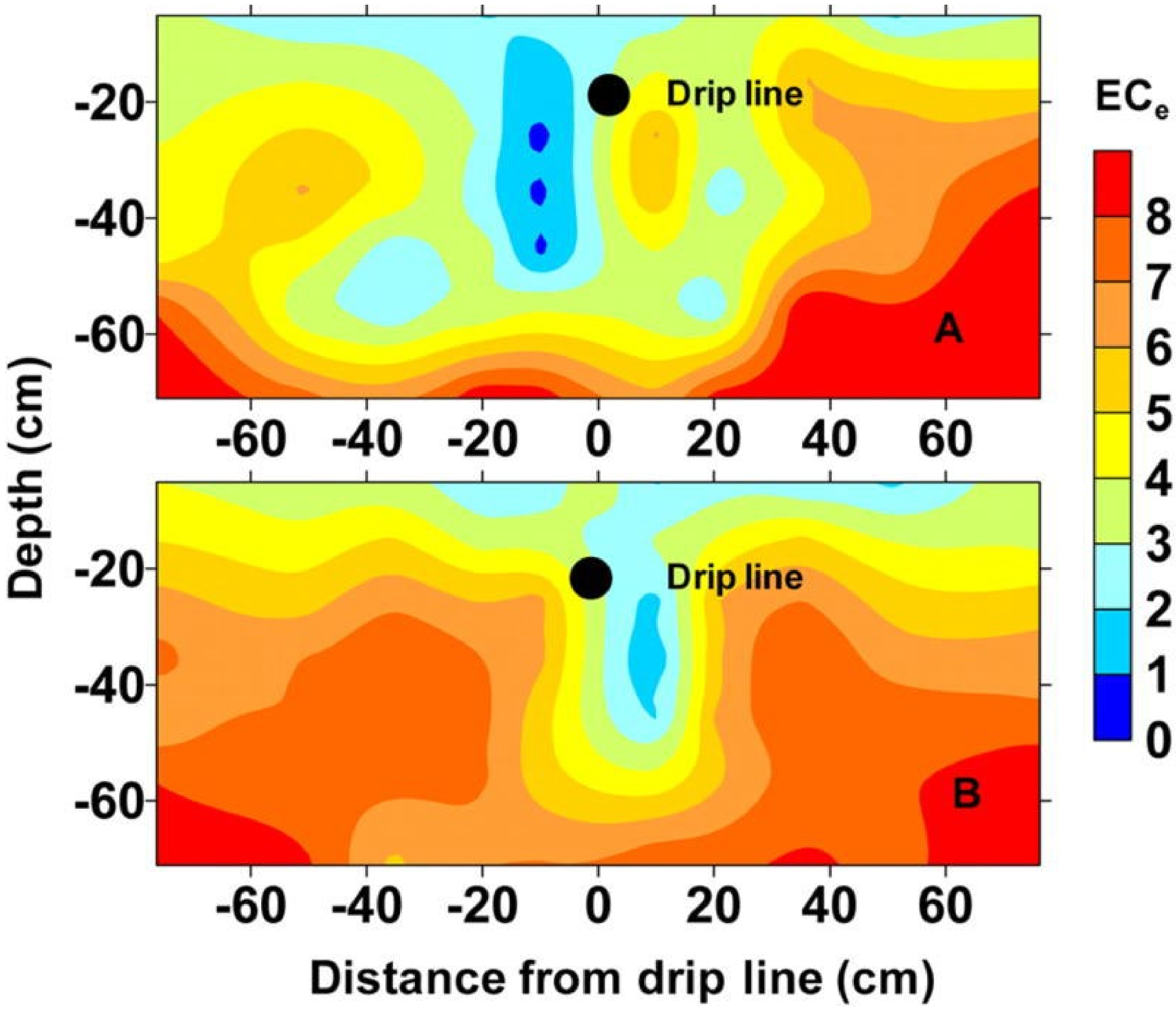

Figure 2-7. Actual Salt Distribution in Subsurface Drip Irrigated Field.
Source: Hanson et al., 2008.

As can be seen from these figures, soil salinity is least where irrigation water enters the soil surface. This is true whether the irrigation water enters the soil by furrow, sprinkler or drip irrigation. Then, as soil water moves away from the point of entry, roots extract water, concentrating the salts along the way. It is this water flow direction and root water extraction that these characteristic salt patterns develop. The low salinity zone, regardless of irrigation method, is where most of the roots will proliferate. Note that where soil salinity was characterized in a subsurface drip irrigated field (Figure 2-6, bottom), the actual salinity distribution is also influenced by the heterogeneity of the soil.

2.2.6.2 Furrow Irrigation and Seed-Bed Management

Investigators have long understood how salts move in soils under different irrigation methods and have developed planting strategies to optimize stand establishment. Yield losses in fields are often attributed to failures in germination and emergence (Hamdy, 1993) largely due to lack of control of salinity in the upper soil layer. Seed bed shape and seed location should be managed to minimize salinity effects (Figure 2-8). For soils irrigated with saline recycled water, sloping beds are the best where seedlings can be safely established on the slope below the zone of salt accumulation (Bernstein and Fireman, 1957). In this configuration, salts move with the soil water past the seedling and either towards a higher portion of the seedbed or in the middle of the bed between emerging plants.

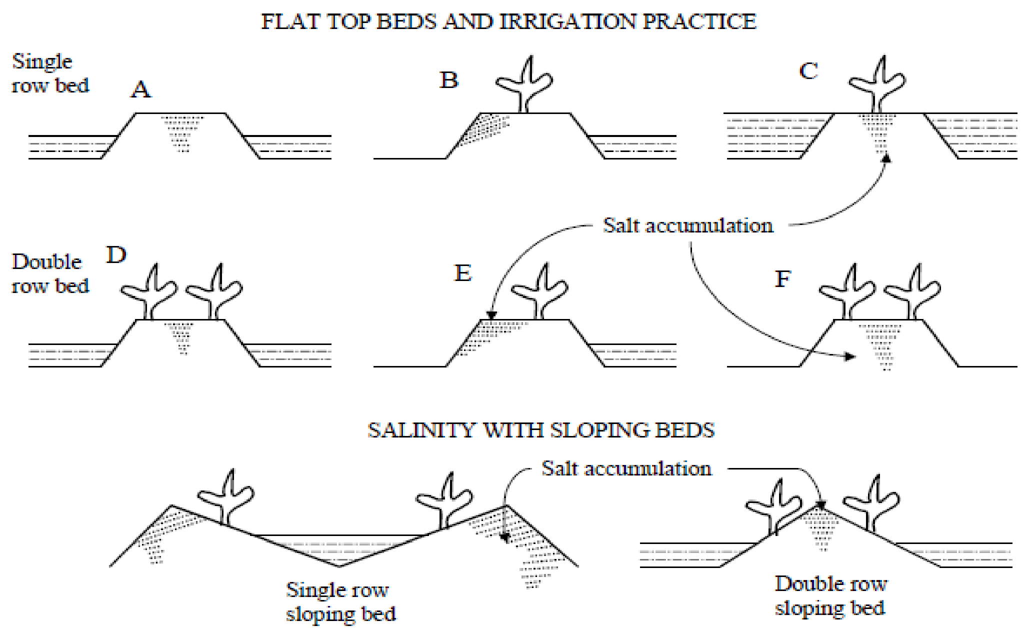

Figure 2-8. Typical Salt Accumulation Pattern in Ridges and Beds Cross Section in Soils Irrigated by Furrows.
Source: Bernstein and Fireman, 1957.

Crop roots will exploit the soil profile in the most favorable conditions of salinity, water content, soil strength, aeration, pH and available nutrients. However, understanding how the plant responds to soil conditions that vary over space and time is very difficult. In regard to irrigation methods, crops typically perform better under irrigation with saline recycled-water using drip irrigation and worse under sprinkler irrigation.

2.2.6.3 Drip Irrigation

Under drip irrigation, the salinity of the soil water near the dripper is close to that of the irrigation water or slightly above. Moreover, a well-designed drip system reduces weed growth, improves distribution uniformity, reduces unnecessary water losses and allows for better fertilizer application (Hanson et al., 1997; Lamm et al., 2006). Because root density is highest where soil conditions are most favorable, crops under drip irrigation can take advantage of this low-salinity zone that does not exist under sprinkler or surface irrigation methods. In addition, with frequent irrigation and controlled application rates, inherent soil heterogeneity throughout the field can be partially overcome than would otherwise with surface irrigation methods. The latter method would lose more water in the sandier or portions of the field with the highest infiltration rates. The main limitations of drip irrigation lie in the higher initial cost, power and water supply needs, and higher management skills to effectively run the system. The development of high soil salinity between drippers requires end of season leaching to avoid potential damage to subsequent crops. There is also the concern that under saline recycled water irrigation that drip emitters will be more vulnerable to chemical clogging. Often, recycled water is slightly alkaline and contains substantial amounts of calcium. Calcite can precipitate on the outside of the emitters reducing the emitter flow rate. Because of this concern, periodic acid injection is recommended to reduce calcite precipitation within the emitters.

2.2.6.4 Sprinkler Irrigation

Sprinkler irrigation allows the irrigator to not only apply the irrigation water uniformly compared with furrow irrigation but also to control the rate of water application. Such an irrigation method is ideal for leaching because salt transport is predominantly downward and pre-plant leaching of the topsoil layer will help with stand establishment. Under sprinkler irrigation, applied water can be controlled at or below the infiltration rate but this method of irrigation typically wets the canopy. Leaves that are wetted by saline sprinkler water can absorb salts directly making them more susceptible to sodium and chloride toxicity (Maas, 1985) (see Figure 2-9). If sprinkler irrigation, however, can be managed to irrigate the field below the canopy and not wet the leaves, such as in orchards below the canopy, crop damage from foliar absorption of salts can be avoided.

Figure 2-9. Increasing Impact of Salt Injury from Sprinkler Irrigation. Leaf on Left from Non-saline Sprinkler Irrigated Plants and the Three on the Right Were Sprinkler Irrigated with 30 Meq/L Salt Solution.
Source: Photo by S. Grattan.

2.2.7 Potential Clogging

Some irrigation methods are more suitable for saline water or other types of marginal- or low-quality water than others (Nakayama, and Bucks, 1991). Drip irrigation systems, for example, have the advantage of reducing the amounts of water loss while reducing salinity impacts. Table 2-3 presents water quality requirements to prevent clogging in localized irrigation systems. Solids in the effluent or biological growth at the emitters will create problems but gravel filtration of secondary treated effluent and regular flushing of lines have been found to be effective in preventing such problems (Nakayama and Bucks, 1991).

Table 2-3. Water Quality and Clogging Potential in Drip Irrigation Systems.
Source: Nakayama and Bucks, 1991.

Potential Problem	Units	Degree of Restriction on Use		
		None	Slight to Moderate	Severe
Suspended Solids	mg/l	< 50	50- 100	> 100
pH		< 7.0	7.0 - 8.0	> 8.0
Dissolved Solids	mg/l	< 500	500-2000	> 2000
Manganese	mg/l	< 0.1	0.1 - 1.5	> 1.5
Iron	mg/l	< 0.1	0.1 - 1.5	> 1.5
Hydrogen sulfide	mg/l	< 0.5	0.5 - 2.0	> 2.0

Managing salinity and water stress simultaneously is a complex challenge. However, salt-stressed crops might not respond positively to increasing irrigation frequency unless it reduces water stress, maintains the salt concentration in the soil solution below growth-limiting levels, and does not contribute to additional stresses such oxygen deficit or root diseases (Maas and Grattan, 1999).

Several benefits of high frequency irrigation do exist, regardless of salinity. These include increased water availability for root uptake, more root activity, and improved nutrient management options. Mineral nutrition has been shown to reduce specific toxicity of salts and thus proper high frequency fertigation could be particularly beneficial for saline conditions (Silber, 2005).

2.3 Future Research Directions

2.3.1 Irrigation Strategies Using Saline Recycled Water

The use of recycled water for irrigation that is saline-sodic requires improved management from standard water management practices such as 1) selecting the appropriate crops and crop rotations, 2) identifying the most appropriate method of irrigation, 3) determining the amount, timing and method of irrigation to achieve the necessary leaching and 4) selecting the type and amount of amendments if soils are also sodic. To sustain good management practices, continuous monitoring of the irrigation water, soils and plants must be conducted to make sure salinity and sodicity are controlled within manageable limits. Most of the scientific foundation for management decisions has been laid out earlier in this chapter however the focus has been on just one source of irrigation water available; recycled water that is saline. Management practices that optimize crop production depend upon whether low salinity water is also available for irrigation. If two sources of water, saline and non-saline, are available for irrigation, then several other irrigation strategies can be considered.

2.3.2 Mixing or Blending Irrigation Waters

When two sources of water are available for irrigation, blending the two in proportions to provide water of suitable quality is an obvious option. The goal of ‘mixing’ is to blend two sources of irrigation water together with the overall goal of achieving a larger volume of water but of suitable quality for irrigation. The suitability of the water depends on the crop tolerance of the crop being irrigated.

The following formula can be used to blend two sources of irrigation water of different qualities. The blending ratio (BR) is the volume of good quality irrigation water applied to the field divided by the volume of saline water applied to the field and is calculated as follows;

$$BR = (EC_s - EC_b) / (EC_b - EC_w) \qquad \text{(Equation 2-2)}$$

where ECw, ECs and ECb are the electrical conductivities of the good quality water, the saline recycled-water and the blended water, respectively. Therefore, crops that are more tolerant can use lower blending ratios. The ECb can be assigned depending upon the crop salt tolerance or acceptable level of yield decline based on targeted leaching and the BR is then calculated knowing the EC of the two different water sources.

Mixing irrigation waters together is a way to expand the amount irrigation water available for irrigation but there are limits on how salty the saline water can be. Blending only expands the usable water supply when the saline water component, if applied independently without blending, can still produce a crop (Grattan and Rhoades, 1990; Rhoades et al., 1992). In other words, the crop can still extract water and grow albeit at a very low rate. The water is 'too salty' if it is applied by itself and kills the crop, regardless of management and leaching. For example, one liter of fresh water mixed with on liter of seawater equals 2 liters of water at half sea-water strength. If onions or rice were the crop selected, then this blended water is too salty and cannot be used to irrigate these crops. In this example, it would be better to use the one-liter fresh water without blending. By blending these waters, one liter of fresh water is effectively lost from the system because it is blended with too salty water.

2.3.3 Cyclic Use of Saline and Non-Saline Water

The cyclic strategy alternates between the use of saline irrigation water and freshwater usually at different times in the growing season and/or for different crops within a crop rotation. Typically, fresh water is used early in the season to reduce soil salinity in the upper profile, facilitating germination and permitting crops with lesser tolerances to salinity to be included in the rotation (Rhoades et al., 1992). Saline water is used for more salt-tolerant crops or for more salt-sensitive crops later in the season.

The objective of the cyclic strategy is to minimize soil salinity (i.e., salt stress) during the salt-sensitive growth stages, or when salt-sensitive crops are grown in a rotation of crops. This does not simply imply that saline recycled water is only applied to salt-tolerant crops after they reach a salt-tolerant growth stage or that fresh water is only used to irrigate salt-sensitive crops. Soil salinization lags behind saline water application, so that it takes time for a soil profile to become salinized. This allows a more salt-sensitive crop to be irrigated with saline water later in the season in conditions where the soil was initially non-saline at the beginning of the season (Shennan, et al., 1995; Bradford and Letey, 1992). Similarly, without pre-plant leaching or sufficient rainfall, it is often difficult to return to a salt-sensitive crop using non-saline water in a soil that was previously salinized.

2.3.4 Comparing Irrigation Strategies

Each method of irrigation with saline water has its advantages and disadvantages. Mixing is the easiest practice while alternating fresh and saline waters requires some knowledge on the varying crop tolerance level during the different growth stages. In addition, mixing requires that both fresh and saline water are always available. Alternating saline and fresh water, on the other hand, offers a better salt leaching mechanism. That is, when saline water irrigation is followed by fresh water, the latter will leach the salts accumulated in the soil from the saline irrigation. This keeps the soil profile in a transient state. Mixing does not offer such possibility as it continuously adds salts so salinity is only controlled by post season leaching. While the 'cyclic' method has advantages over the 'blending' method, it requires a higher level of management skill to make this practice sustainable.

Irrigation with saline-sodic recycled water requires a higher level of management over the long term, than does irrigation with non-saline water, not only to avoid long-term salinization but to maintain soil physical conditions. Soil physical properties can be affected by irrigation with saline-sodic water, particularly when good quality water is used or rainfall occurs after saline-sodic water application (Grattan et al., 2012). These adverse effects include reduced infiltration, poor soil tilth, and poor aeration resulting in anoxic conditions in the root zone (Oster and Shainberg, 2001). These negative impacts can be minimized with amendments like gypsum, sulfur, and sulfuric acid being applied to either the soil or irrigation water (Oster et al., 1992). In addition, if high levels of B are present in the water, its accumulation in the soil could adversely affect crop production (Grattan and Oster, 2003). Boron is particularly problematic in that, as stated earlier, it roughly takes three times the amount of irrigation water to reclaim that soil than it does to reclaim saline soil.

The need to leach salts and B from the root zone will also leach nitrate. Nitrate losses can be mitigated by additional fertilizer application, but such losses are environmentally damaging and economically unwise. On the other hand, if saline recycled water contains substantial amounts of NO_3^-, some crops can be adversely affected while other crops can benefit (Kaffka et al., 1999). That is, excess nitrate in the soil water late season can induce excessive vegetative growth and produce poor quality crops. This has been observed in grapes and processing tomatoes. And high late-season NO_3^- can reduce the sugar content in sugar beet. But NO_3^- can be beneficial to crop production by reducing the amount of fertilizers that need to be applied.

2.3.5 Sequential Use of Saline Water

Sequential use of saline water is applicable in fields with drain lines installed to collect the drainage water to help control the level of the perched water table. In this practice, the farm is divided into a conventional irrigation area and an area where recycled water is used for irrigation. The conventional portion of the field contains high value, salt sensitive crops that are irrigated with low saline water. The saline recycled area consists of a sequence of fields that are irrigated with saline water of increasingly higher concentrations (see Grattan et al., 2012). That is, the drainage water is collected under fields planted with conventional crops which is more saline than the irrigation water. This drainage water is then used to irrigate the next field (crops of higher salt tolerance) in the sequence where the volume of drainage water decreases and the salinity increases (Figure 2-10). The process then continues to the next field. The main purpose is to obtain an additional economic benefit from the available water resources, minimize the area affected by shallow water tables and reduce the volume of drainage water that requires disposal.

Although sequential reuse is a conceptually attractive means of recycling drainage water on a farm or at a district level, there is a large lag time for salts at the beginning field to reach the final stage of the sequence. Using a transfer function model, assuming typical drain-line spacing and water management practices, investigators found that such a reuse system would never effectively reach steady-state, but rather it could take decades or much longer for water and dissolved salts to move though the sequential system (Jury et al., 2003). In addition, the salt removal via harvesting of salt tolerant and halophytic plants represents a very small fraction of salt removed from the sequential system. Therefore, caution is advised for those designing sequential reuse systems and estimating the rate of salt movement through the sequential system, particularly if steady-state assumptions are used (Grattan et al., 2014; Hopmans et al., 2021). Drainage water reuse systems are subjected to fluctuating water tables, due to off-farm conditions. These fluctuations, particularly where the water table depth is below the tile lines, will also affect the time needed to establish quasi-steady-state conditions.

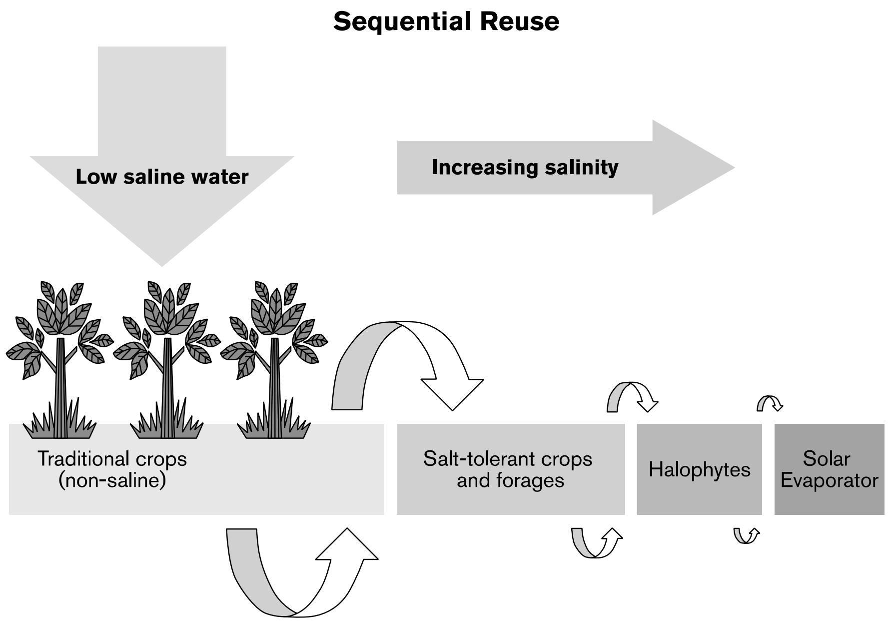

Figure 2-10. Sequential Use of Saline Water Where Drainage Water Is Collected and Reused on Progressively More Salt Tolerant Crops before the Final Concentrated Drainage Water is Evaporated in a Solar Evaporator.
Source: Grattan, S.R., J.D. Oster, J. Letey, and S.R. Kaffka; "Drainage Water Reuse: Concepts, Practices, and Potential Crops. Salinity and Drainage in San Joaquin Valley, California." *Global Issues in Water Policy*, (5): 277-302, 2014. Springer Nature.

2.3.6 Soil Amelioration

Specific soil ameliorants can alter the crop availability of micro-nutrients. Liming with $CaCO_3$, for example, can increase soil pH from 5.5 to 7.0 which results in a significant reduction in cadmium uptake of many crops (Gray et al., 1999; Zhu et al., 2016). Other materials, such as organic waste, sawdust, or biochar can absorb heavy metals from irrigation water. In the case of irrigation with sodic waters or managing soils with a high ESP, there is a need to provide a source of calcium to mitigate the effects of sodium and in certain cases of magnesium on soils and crops.

Gypsum application techniques have been refined in the form of 'gypsum beds', the use of which improves gypsum's solubility and application efficiency and reduces the costs of its application. Although this method produces significantly higher crop yields than any control, it can be constrained in many developing countries because of (1) low quality (impurities) of available gypsum; (2) restricted availability of gypsum, in absolute terms or when actually needed; and/or (3) increased costs due to competing demands for it (Qadir et al. 2007).

Another low-cost source of calcium is phosphogypsum, which can be used as an amendment for managing high-magnesium waters and soils. Phosphogypsum is a major co-product of the production of fertilizer from phosphate rock. Where phosphate rock is available and mined, phosphogypsum offers additional value as it also supplies some phosphorus and sulfur essentially needed for plant growth (Vyshpolsky et al., 2008).

As for contaminants, crop residues, municipal waste compost, manure or biochar can also be useful in ameliorating the effects of soil and irrigation water sodicity. The organic matter left in or added to the field can improve the chemical and physical conditions of the soils irrigated with sodic wastewater by supporting the dissolution of calcite due to enhanced CO_2 production from the microbial breakdown of organic matter (Leogrande and Vitti 2019).

2.3.7 Phytoremediation

Phytoremediation could be a preferable option for amending soils in developing countries as it is inexpensive and easily scalable nature. This technique is based on the use of living green plants to fix, adsorb or dissolve contaminants or salts.

Phytoremediation in a more scientific way might refer to a process internal or external to the plant: (a) the ability of plant roots to absorb particular ions for the plant to accumulate them, or (b) chemical changes in the root zone (partial pressure of carbon dioxide increase which influences the dissolution rate of calcite), resulting in enhanced levels of Ca^{2+} in the soil solution to possibly replace Na^+ from the cation exchange complex depending on respective available amounts (Qadir et al., 2007).

While the first option is more popular in view of trace elements and some heavy metals the second option can be effective when used on moderately saline-sodic and sodic soils if soluble calcite and appropriate plant species are available. On highly sodic and saline-sodic soils, use of chemical amendment is likely to outperform phytoremediation treatments (Qadir et al., 2007).

2.3.8 Crop Diversification

A pertinent selection of plant species capable of withstanding ambient levels of salinity and/or sodicity and producing adequate yields is crucially important for using saline, sodic, or saline-sodic waters for irrigation. Such restrictions are generally limited to the possibility of access to a replacement crop (or variety) with financially viable market value.

2.3.9 Forage Grass and Shrub

Promising forage species as reported by different researchers include, but not limited to, tall wheatgrass, Kallar grass, Para grass, Bermuda grass, Kochia, sesbania, purslane, and shrub species from the genera *Atriplex* and *Maireana* (Barrett-Lennard, 2002; Robinson et al., 2003).

2.3.10 Bio-Energy Crops

Several studies have shown that a range of plant species can be used for renewable energy production in salt-affected environments. Some promising examples are jatropha, toothbrush tree, Russian olive, and sweet-stem sorghum (Qadir et al., 2010).

Several **fruit-tree species** have shown promising results under saline environments. The prominent fruit trees for saline environments are date palm, olive, chicle, guava, Indian jujube, and karanda (Qureshi and Barrett-Lennard, 1998). Studies on establishing rapidly growing **tree plantations** can offer an opportunity of using salt-affected lands to provide firewood under saline environments, using a variety of tree indigenous and exotic species (Qadir et al., 2008; Qureshi and Barrett-Lennard, 1998). The selection of tree species for salt-affected lands usually depends on the cost of inputs and the subsequent economic and/or on-farm benefits.

2.4 Summary

Irrigation with saline-sodic recycled water requires a higher level of management over the long term, than does irrigation with non-saline water, not only to avoid long-term salinization but to maintain soil physical conditions. Soil physical properties can be affected by irrigation with saline-sodic water, particularly when good quality water is used or rainfall occurs after saline-sodic water application (Grattan et al., 2012). These adverse effects include reduced infiltration, poor soil tilth, and poor aeration resulting in anoxic conditions in the root zone (Oster and Shainberg, 2001). These negative impacts can be minimized with amendments like gypsum, sulfur, and sulfuric acid being applied to either the soil or irrigation water (Oster et al., 1992). In addition, if high levels of B are present in the water, its accumulation in the soil could adversely affect crop production (Grattan and Oster, 2003). Boron is particularly problematic in that it roughly takes three times the amount of irrigation water to reclaim that soil than it does to reclaim saline soil. Management practices such as blending, cycling and sequential use should be adopted when saline-sodic recycled water is used for irrigation.

CHAPTER 3

Heavy Metals and Recycled Water Irrigation

3.1 Introduction

Pollution of agricultural environments by heavy metals has been a constant concern for decades. Natural factors (erosion, atmospheric deposits, volcanic activities, etc.), different anthropogenic activities (irrigation with sewage, addition of manure, fertilizers, etc.), together with high rates of mobilization and transport, have accelerated the contamination process (Khan et al., 2004; Khan et al., 2013; Sen Gupta et al., 2020).

As compared with other pollutants, heavy metals are non-biodegradable, and so they persist for a long time in the environment (Ali et al., 2019), contaminating the food chain, and causing different health problems due to their toxicity. In addition, chronic exposure to heavy metals in the environment is a real threat to living organisms (Wieczorek-Dąbrowska et a., 2013).

Heavy metals and metalloids in agricultural environments include Cr, Ni, Cu, Zn, Cd, Pb, Hg, or As. Specifically, Chaney et al. (1980) classified metals into four groups according to the health risks that metals can produce when they are introduced in the food chain. Group 1 is comprised of the elements Ag, Cr, Sn, Ti, Y and Zr, which pose little risk because they are not taken up to any extent by plants, owing to their low solubility in soil and, consequently, negligible uptake and translocation by plants. Elevated concentrations of these elements in foods usually indicate direct contamination by soil or dust. Group 2 includes the elements As, Hg, and Pb which are strongly sorbed by soil colloids, and while they may be absorbed by plant roots, they are not readily translocated to edible tissues, and therefore pose minimal risks to human health. As a soil contaminant, fluorine would also fall into Group 2, but as an atmospheric contaminant, F may be readily absorbed by plants and could pose localized food-chain health risks. Group 3 is comprised of the elements B, Cu, Mn, Mo, Ni, and Zn, which are readily taken up by plants, but are phytotoxic at concentrations that pose little risk to human health. Conceptually, the `soil–plant barrier' protects the food chain from these elements. Group 4 consists of Cd, Co, and Se, which pose human or animal health risks at plant tissue concentrations that are not generally phytotoxic.

The quality of the recycled water can contribute to the number of heavy metals in agricultural soils. Barbieri, (2016) reported that heavy metal concentrations above threshold levels affected the microbiological balance of soils and reduced soil fertility. Plants can also uptake heavy metals, Table 3-1 shows the form of various heavy metals available for plant uptake and their effect on plant growth and productivity.

Table 3-1. The Form of Heavy Metal Available for Uptake and Plant Effects.

Heavy Metal	Form Available to Plant Uptake	Plant Metabolism	Plant Effects
Cd	Cd^{+2}	• Non-essential element	• Photosynthesis • Water uptake and nutrient uptake • Necrotic cell death • Uncontrolled cell proliferation • Formation of reactive oxygen species
As	As^{5+}	• Non-essential element	• Metabolic processes of phosphate
Pb	Pb^{2+} and lead-hydroxy complexes	• Non-essential element	• Seed germination • Plant growth • Chlorophyll synthesis
Cr	Cr^{+3} Cr^{+6}	• Non-essential element	• Seed germination • Plant growth • Chlorophyll synthesis • Seedling dry matter • Development of stems and leaves during plant early growth stage • Cell division • Elongation of roots and shoot • Metabolic disorders in seed germination
Zn	Zn^{2+}	• Component of special proteins known as zinc fingers that bind to DNA and RNA • Constituent of enzymes (oxidoreductases, transferases, Hydrolases) and ribosome	• Photosynthesis • Genetic-related disorders • Plant growth • DNA regulation and stabilization • Necrotic spotting
Ni	Ni^{2+}	• Key factor in activation of enzyme urease (nitrogen metabolism) • Seed germination • Iron uptake	• Seed germination and seedling growth by hampering the activity of the enzymes (amylase and protease) • Plant height and leaf area • Inhibition of lateral root formation and development • Damages in photosynthetic apparatus • Hamper mitotic cell division in root
Cu	Cu^{2+}	• Catalyzer of redox reaction in mitochondria, chloroplasts, and cytoplasm of cells • Electron carrier during plant respiration	• Root elongation and growth • Lipid peroxidation • Structural disturbance of thylakoid membranes • Cell elongation • Seedling growth • Photosynthesis

3.2 Remediation Techniques for Heavy Metals in Agricultural Environments

The prevention and amendment of heavy metal concentrations in agricultural environments are not only necessary to control the sources (irrigation wastewaters), but also to enhance the remediation efficiency of contaminated soils. Viable risk reduction options in both irrigation wastewater and contaminated soils can be classified according to Table 3-1. The choice of appropriate wastewater or soil treatment depends primarily on the degree of contamination and the costs associated and not all of which are affordable for developing countries.

The phytoremediation technique can be considered a preferable option for amended soils in developing countries due to its being inexpensive and easily applicable to large sites and quantities of contaminants derived from various sources. This technique is based on the use of living green plants to fix or adsorb contaminants for cleaning soils, reducing chemical risks. Its advantages include its applicability to a wide variety of pollutants while minimizing the generation of secondary waste. In addition, it is profitable for large areas, leaving the top layer of the soil in conditions usable for agriculture. Actually, hyperaccumulators are recommended for phytoremediation due to its metal retention capacity of up to 100 times higher than other species. Van der Ent et al., (2013) recommended the following concentration criteria for different metals and metalloids in dried foliage with plants growing in their natural habitats: a) 100 mg kg^{-1} for Cd, Se and Tl; b) 300 mg kg^{-1} for Co, Cu and Cr; c) 1000 mg kg^{-1} for Ni, Pb and As; d) 3000 mg kg^{-1} for Zn; 10,000 mg kg^{-1} for Mn.

However, when the concentration of contaminants is too high, soil replacement, soil removal, or soil isolation may be necessary. All of these methods are associated with large amounts of manpower and material resources, so they can only be applied to small areas. In addition, it would be necessary to apply a later technology to try to treat the soil that has been removed.

3.3 Effects of Chronic Exposure to Heavy Metals through Consuming Wastewater-Irrigated Food

The presence or excessive accumulation of chemicals in agricultural systems through recycled water irrigation may not only result in soil contamination but also affect food quality and safety due to the capacity of certain elements or compounds to transfer to the food chain via irrigation water → soil → plant → human route. In particular, continuous exposure to certain persistent organic chemicals or heavy metals including arsenic, cadmium, lead, or mercury due to prolonged consumption of contaminated foods is linked to a wide range of chronic health effects. Some of these compounds have cumulative tendencies in the human body through food exposures, leading to short-term and long-term health problems such as the decline of essential nutrients, deficiencies of the central nervous system or cardiovascular, gastrointestinal, haematological, hepatic, renal, neurological, developmental, reproductive, and immune diseases (Table 3-2).

Table 3-2. Behaviour of Organic Compounds and Heavy Metals in Humans.
Source: Adapted from Goyer, R. 2004 (US EPA) and EPA, U. (2007).

Body Effects	Organic Chemicals	Heavy Metals
Metabolism	Generally extensive and often species-specific.	Usually limited to oxidation state transitions and alkylation/dealkylation reactions.
Persistence	Common in body fat due to lipid solubility.	Possibility of binding to plasma proteins or tissue proteins.
Removal	By excretion in urine after biotransformation from lipophilic forms to hydrophilic forms, in bile after conjugation to large organic molecules, or in exhaled air if not metabolized.	In urine because metal compounds are generally small molecules and are hydrophilic. As a result of protein binding, may be excreted via hair and fingernails.
Tissue uptake	Commonly a blood flow-limited process, with linear portioning into tissues.	Metals and their complexes are often ionized, with tissue uptake (membrane transport) having greater potential to be diffusion-limited or to use specialized transport processes.
Interactions	May occur, especially during metabolism.	Commonly during the processes of absorption, excretion, and sequestration.

3.4 Health Risk Assessment Indices

Risk assessment in irrigated crops with contaminated water focuses primarily on soil adsorption capacity and soil-plant transfer, determined by the bioaccumulation factor (BAF). Although irrigation water may have low levels of chemicals, long-term irrigation with wastewater containing potentially toxic chemical compounds can lead to the accumulation of undesirable compounds in soils, being the main route of exposure to humans through the consumption of food crops. In order to assess the magnitude and develop numerical limits of potential risks to human health associated with the exposure to chemicals, the following factors should be considered:

- Hazard identification (identification of toxic chemicals)
- Dose-response evaluation and risk characterization (maximum permissible exposure levels for each chemical, based on the dose-response characteristics associated with a predetermined acceptable risk level).
- Exposure analysis (focusing to realistic scenarios)

The major effects of recycled water on crop productivity are due to the presence of heavy metals, which are well-known to negatively affect crop productivity. Under this perspective, information about specific type of heavy metal present in food crops and their dietary intake is necessary for assessing their risk to human health. Useful parameters for the evaluation of risks associated with the consumption of heavy metal contaminated food crops are described in Table 3-3. Although these parameters were specifically defined for the assessment of risk to heavy metals, they are currently being used to assess risks to other relevant organic contaminants such as contaminants of emerging concern (CECs).

Table 3-3. Soil-Plant Heavy Metal Transfer and Health Risk Assessment Indices.

Index	Application	Formula		Comments	References
Bioaccumulation, bioconcentration or transfer factor (BAF, BCF or TF)	Uptake capacity of metals from soils to plants.	$BAF = \frac{C_{plant\ tissue}}{C_{soil}}$	$C_{plant\ tissue}$: Concentration of metal in plant tissue (root, stem or leaves), (mg/kg) C_{soil}: Concentration of metal in soil (mg/kg)	BAF >1 higher accumulation of metals in plant parts than soil	Oti, W. O. (2015) Islam et al., (2016) Ghasemidehkordi et al. (2018) Rai et al., (2019)
Estimated daily intake (EDI) or Daily Intake of Metal (DIM)	Assess the relative phyto-availability of metal	$EDI = \frac{C_{metal} x\ W_{food}\ x\ Cf}{B_w}$	C_{metal}: Conc. of metals in crops (mg/kg) W_{food}: Daily average intake of vegetables (kg/day) Bw: Body weight (60 kg for adult residents, FAO 2006) Cf: Conversion factor: 0.085 to convert fresh weight to dry weight		Qureshi et al., (2016) Rai et al., (2019)
Daily dietary index (DDI)	Assessment of daily intake of metals through vegetables	$DDI = \frac{C_{metal}\ x\ W_{food} x\ D}{B_w}$	D: Dry weight of the vegetable taken (kg)		Rai et al., (2019)
Hazard quotient (HQ)	Assessment of metals risks to human health	$HQ = \frac{D_w * C_{metal}}{RfD * B_w}$	Dw: Dry weight of consumed vegetable (mg/day) RfD: Oral reference dose for the metal (mg/Kg/day)		Khan et al., (2009)
Hazard Index (HI)	Potential risks for more than one metal.	$HI = \sum HQ$	$\sum HQ = HQ_{metal1} + HQ_{metal2} ...$	The magnitude of adverse effect will be proportional to the sum of multiple metal exposures.	Akter et al.,(2019) Patrick-Iwuanyanwu et al., (2017)
Health Risks Index (HRI)		$HRI = \frac{EDI}{RfD}$		HRI<1 Safe levels HRI≥1 Potential health risks	Rai et al., (2019)
Target Hazard Quotient (THQ)	Assessment of health risks through consumption of vegetables by the local inhabitants	$THQ = \frac{E_F x\ E_D x\ F_{IR} x\ C_{metal}}{RfD\ x\ B_W\ x\ T_A}$	E_F: Exposure frequency (365 days/year), E_D: Exposure duration (70 years), F_{IR}: Food ingestion rate (g/person/day), TA: Average exposure time for non-carcinogens (365 days/year x 70 years)	THQ < 1 Safe levels THQ >1 Adverse effects on the exposed population	Chien et al., (2002) Zhuang et al., (2009)

3.5 Removal of Heavy Metals by Agricultural Waste Products

Considering the economic limitations that developing countries may face with regards to dealing with heavy metal pollution, new trends in viable risk reduction options for wastewaters using low-cost agricultural waste products have been proposed. In this case, for example the use of dairy manure compost as adsorbent has been shown to have maximum adsorption capacities of 15.5, 27.2, and 95.3 mg g^{-1} for Zn(II), Cu(II), and Pb(II), respectively (Zhang, 2011), while residual waste materials from rice (rice bran, rice straw, or rice husk) have demonstrated the partial reduction of Cu (II), Zn (II), Cd (II), Mn (II) or even Pb (II). (Singha and Das, 2013; Krishnani et al., 2008). Other options for heavy metal removal are based on the use of some mineral deposits, or aquatic and terrestrial biomass. Some examples of the use of low-cost materials for the reduction of heavy metals in wastewater are described in Table 3-4.

Table 3-4. Removal of Heavy Metals by Various Agricultural Waste Products.

Adsorbent	Heavy Metals	Surface Area (m^2 g^{-1})	Conc. (mg L^{-1})	qmax [1](mg g^{-1})	Reference
A. Hypogea (peanut) *shells*	Chromium (VI)	1.8	0-40	4.3	Ahmad et al. (2017)
Almond	Chromium (VI)	N.R.	20-1000	10.2	Dakiky et al. (2002)
Apple residues	Copper (II)	N.R.	30	10.8	Lee and Yang (1997)
Banana peel	Cadmium (II)	1.3	100-800	34.1	Thirumavalavan et al. (2010)
	Cobalt (II)	N.R.	5-25	2.6	Annadural et al. (2002)
	Copper (II)	1.3	100-800	52.4	Thirumavalavan et al. (2010)
	Copper (II)	N.R.	5-25	4.8	Annadural et al. (2002)
	Copper (II)	38.49	10-30	7.4	DeMessie et al. (2015)
	Lead (II)	1.3	100-800	25.9	Thirumavalavan et al. (2010)
	Lead (II)	N.R.	5-25	7.9	Annadural et al. (2002)
	Nichel (II)	1.3	100-800	54.4	Thirumavalavan et al. (2010)
	Nichel (II)	N.R.	5-25	6.9	Annadural et al. (2002)
	Zinc (II)	1.3	100-800	21.9	Thirumavalavan et al. (2010)
	Zinc (II)	N.R.	5-25	5.8	Annadural et al. (2002)
Cashew nut shells	Copper (II)	395	10-50	20.0	Senthilkumar et al. (2011)
	Nichel (II)	395	10-50	18.9	Senthilkumar et al. (2011)
Coconut-shell	Chromium (VI)	0.5	54.5	18.7	Singha and Das (2011)
	Copper (II)	N.R.	5-300	19.9	Singha and Das (2013)
Coconut-shell biochar	Cadmium (II)	212	100-2000	3.5	Paranavithana et al. (2016)
	Lead (II)	212	100-2000	13.4	Paranavithana et al. (2016)
Corncob	Cadmium (II)	<5	5-120	5.1	Leyva-Ramos et al. (2005)
	Lead (II)	N.R.	20.7-414	16.2	Tan et al. (2010)
Dairy manure compost	Copper (II)	N.R.	31.8	27.2	Zhang (2011)
	Lead (II)	N.R.	103.6	95.3	Zhang (2011)
	Zinc (II)	N.R.	32.7	15.5	Zhang (2011)

Adsorbent	Heavy Metals	Surface Area ($m^2 g^{-1}$)	Conc. ($mg L^{-1}$)	qmax [1]($mg g^{-1}$)	Reference
Grapefruit peel	Cadmium (II)	N.R.	50	42.1	Torab-Mostaedi et al. (2013)
	Nichel (II)	N.R.	50	46.1	Torab-Mostaedi et al. (2013)
Grape stalks	Copper (II)	N.R.	15.3-153	10.1	Villaescusa et al. (2004)
	Nichel (II)	N.R.	14.1-141	10.6	Villaescusa et al. (2004)
Groundnut shells	Copper (II)	N.R.	73-465	4.5	Shukla and Pai (2005)
	Nichel (II)	N.R.	107-554	3.8	Shukla and Pai (2005)
	Zinc (II)	N.R.	38-244	7.6	Shukla and Pai (2005)
Hazelnut shells	Copper (II)	441.2	25-200	58.3	Demirbas et al. (2009)
Lemon peel	Cadmium (II)	1.3	100-800	54.6	Thirumavalavan et al. (2010)
	Copper (II)	1.3	100-800	70.9	Thirumavalavan et al. (2010)
	Lead (II)	1.3	100-800	37.9	Thirumavalavan et al. (2010)
	Nichel (II)	1.3	100-800	80.0	Thirumavalavan et al. (2010)
	Zinc (II)	1.3	100-800	27.9	Thirumavalavan et al. (2010)
Orange peel	Cadmium (II)	N.R.	50-1200	293	Feng et al. (2011)
	Cadmium (II)	2.0	100-800	41.8	Thirumavalavan et al. (2010)
	Cobalt (II)	N.R.	5-25	1.8	Annadural et al. (2002)
	Copper (II)	N.R.	5-25	3.7	Annadural et al. (2002)
	Copper (II)	2.0	100-800	63.3	Thirumavalavan et al. (2010)
	Lead (II)	N.R.	5-25	7.8	Annadural et al. (2002)
	Lead (II)	N.R.	50-1200	476	Feng et al. (2011)
	Lead (II)	0.21	57	27.9	Abdelhafez and Li (2016)
	Lead (II)	2.0	100-800	27.1	Thirumavalavan et al. (2010)
	Nichel (II)	N.R.	5-25	6.0	Annadural et al. (2002)
	Nichel (II)	N.R.	50-1200	162	Feng et al. (2011)
	Nichel (II)	2.0	100-800	81.3	Thirumavalavan et al. (2010)
	Zinc (II)	N.R.	5-25	5.3	Annadural et al. (2002)
	Zinc (II)	2.0	100-800	24.1	Thirumavalavan et al. (2010)

[1]Adsorption Capacity

3.6 Summary

Compared with other pollutants, heavy metals are non-biodegradable, and so they persist for a long time in the environment (Ali et al., 2019). The quality of the recycled water can contribute to the number of heavy metals in agricultural soils, affecting the microbiological balance of soils and reducing soil fertility (Barbieri, 2016). Continuous exposure to certain persistent organic chemicals or heavy metals including arsenic, cadmium, lead, or mercury due to prolonged consumption of contaminated foods is linked to a wide range of chronic health effects. In order to estimate the risk, Health Risk assessment in irrigated crops with contaminated water focuses primarily on soil adsorption capacity, and soil-plant transfer and is determined by the bioaccumulation factor (BAF), taking into account the three main factors: Hazard identification (identification of toxic chemicals), Dose-response evaluation and risk characterization (maximum permissible exposure levels for each chemical, based on the dose-response characteristics associated with a predetermined acceptable risk level), and the exposure analysis (focusing on realistic scenarios). For example, according to the USA EPA, the maximum allowable concentrations of heavy metals in recycled water used for irrigation for selected metals are Arsenic (As): 0.01 milligrams per liter (mg/L), Cadmium (Cd): 0.01 mg/L, Chromium (Cr): 0.1 mg/L, Copper (Cu): 0.2 mg/L, Lead (Pb): 0.15 mg/L, Mercury (Hg): 0.002 mg/L, Nickel (Ni): 1.0 mg/L, Selenium (Se): 0.01 mg/L, and Zinc (Zn): 2.0 mg/L.

The phytoremediation technique can be considered a preferable option for heavy metal-amended soils in developing countries due to its being inexpensive and easily applicable to large sites and quantities of contaminants derived from various sources. There are emerging trends in risk reduction from using wastewater by using low-cost agricultural waste products such as dairy manure compost, or aquatic and terrestrial biomass. However, when the concentration of contaminants is too high, soil replacement, soil removal, or soil isolation may be necessary.

CHAPTER 4

Contaminants of Emerging Concerns in Recycled Water used for Irrigation

4.1 Introduction

While the use of recycled water presents multiple economic and environmental benefits, the broad agricultural implementation introduces numerous trace organic contaminants into the agroecosystems (Kinney et al., 2006a; Du et al., 2012; Kostich et al., 2014; Christou et al., 2016). Those contaminants, known as contaminants of emerging concern (CECs), can refer to various synthetic or naturally occurring chemicals, including, but not limited, to pharmaceuticals and personal care products (PPCPs), brominated flame retardants (BFRs), plasticizers, and per- and polyfluoroalkyl substances (PFASs) (Kolpin et al., 2002; Boyd et al., 2003; Kinney et al., 2006b; Fatta-Kassinos et al., 2011; USGS, 2016). The use of everyday products including medicine, cleaning agents, plastics, and other lifestyle products results in the release of thousands of CECs into our wastewater. Since wastewater treatment plants (WWTPs) are only partially effective in removing some of these contaminants, many CECs can pass through and reach farmland through irrigation or application of biosolids, leading to accumulation in soil or even contamination to groundwater (Durán-Alvarez et al., 2009; Siemens et al., 2008; Gibson et al., 2010; Lesser et al., 2018). A number of studies have shown that plants can take up and accumulate CECs in roots, and some of CECs may be further translocated to other plant organs such as leaves and fruits (Goldstein et al., 2014; Wu et al., 2015; Miller et al., 2016; Hurtado et al., 2016; Christou et al., 2017). In addition, in scenarios of irrigation or recurring biosolids applications, CECs may be considered as pseudo-persistent contaminants as they are continually introduced into the soil-plant system at low levels. The potential bioaccumulation and the ensuing environmental health implications of CECs has garnered significant public concerns, as low exposure to certain CECs may lead to unintended adverse consequences.

To date, a significant number of studies have been reported on the occurrence, uptake, environmental processes, and influencing factors of CECs in terrestrial systems and their offsite transport. While the presence of CECs in recycled water is perceived as a potential barrier to the beneficial use of treated wastewater, a pertinent critical review is not currently available. This review synthesizes interrelated processes in the soil-plant system, including sorption, degradation, runoff, and leaching in soil, and uptake, metabolism and translocation by plants, to understand behaviors and fate of CECs in the agroecosystem as a result of recycled water irrigation, and discuss strategies for the mitigation of human and environmental risks. Furthermore, the review identifies the potential challenges and opportunities for the future research on CECs in the context of reuse of recycled water for irrigation.

4.2 Past and Current Knowledge

4.2.1 CECs in Treated Wastewater

A huge amount of synthetic and natural chemicals are consumed annually worldwide, some of which are emitted into the environment directly or indirectly (Petrie et al., 2015). Once consumed by the population and aggregated in sewer lines, CECs flow to wastewater treatment plants (WWTPs), which are only partially effective in their removal. The removal rate of CECs at WWTPs, averaging around 60%, depends on the treatment conditions, input loads, and the chemical's physicochemical properties and resistance to degradation (Kinney et al., 2006a; Margot et al., 2015). Nonpolar compounds are mainly partitioned into sewage sludge while water-soluble polar chemicals remain in the aqueous phase. Persistent polar compounds, such as PFASs, stable benzotriazoles, and some PPCPs, are likely to end up in wastewater effluents due to the insufficient retention times at WWTPs (Helmecke et al., 2020). For example, Golovko et al. (2021) monitored 164 CECs including PPCPs, PFASs, and pesticides in the effluent of 15 WWTPs in Sweden, and reported the presence of 119 CECs in at least one sample, with mean concentrations ranging from 0.11 ng/L to 64,000 ng/L. Similar levels were reported by Vidal-Dorsch et al. (2012) in a southern California study that considered the occurrence of 56 CECs in treated municipal effluents, with average concentrations ranging from ng/L to mg/L. Limited by analytical capabilities and availability of authentic standards, only a few hundreds of CECs have been identified in WWTP effluents to date, merely a small subset of all CECs (Petrie et al., 2015). Furthermore, chemical and microbial actions at WWTPs lead to the formation of many metabolites, which further increases the number of CECs in WWTP effluents (Ferrando-Climent et al., 2012; Fries et al., 2016).

Although the concentrations of CECs in treated wastewater are generally very low, the uncertainty in their potential chronic effects represents a serious concern to sensitive species in ecosystems and human health (Fent et al., 2006). Previous studies have suggested that some CECs affect endocrine systems, especially reproductive health, imposex in molluscs, cancers, various diseases in humans, and freshwater biodiversity (Krishnan et al., 2003; Guillette Jr et al., 1994; Purdom et al., 1994; Jobling et al., 1995; Kidd et al., 2007; Titley-O'Neal et al., 2011; Fowler et al., 2012; Vrijheid et al., 2016; Cacciatore et al., 2018). As most monitoring programs only include a small subset of CECs, any assessment is likely to underestimate the actual risk.

4.2.2 Fate and Behavior of CECs in the Soil-Plant System

A great number of studies have addressed environmental fate of CECs in aquatic environments and during wastewater treatment processes, but relatively little is known about their behavior in the soil-plant system (Anderson et al., 2010; Kalavrouziotis, 2017; Lofrano et al., 2020; Golovko et al., 2021). After entering agricultural soil, CECs are subject to several interrelated processes including degradation and adsorption, which determines the potential for offsite transport (leaching and runoff), as well as the availability of CECs for plant uptake. Once in plants, CECs as xenobiotics may further undergo translocation and metabolism (Figure 4-1) (Rabølle and Spliid., 2000; Kreuzig and Höltge,2005; Boxall et al., 2006; Blackwell et al., 2007; Dolliver et al., 2007; Sherburne et al., 2016). The various fate processes of CECs in soil-plant systems are closely interrelated and driven by soil physiochemical and biological properties, plant species and growth stages, and environmental conditions, management practices such as irrigation system which will ultimately determine the spatial-temporal distribution, bioavailability, offsite movement, and plant accumulation of CECs.

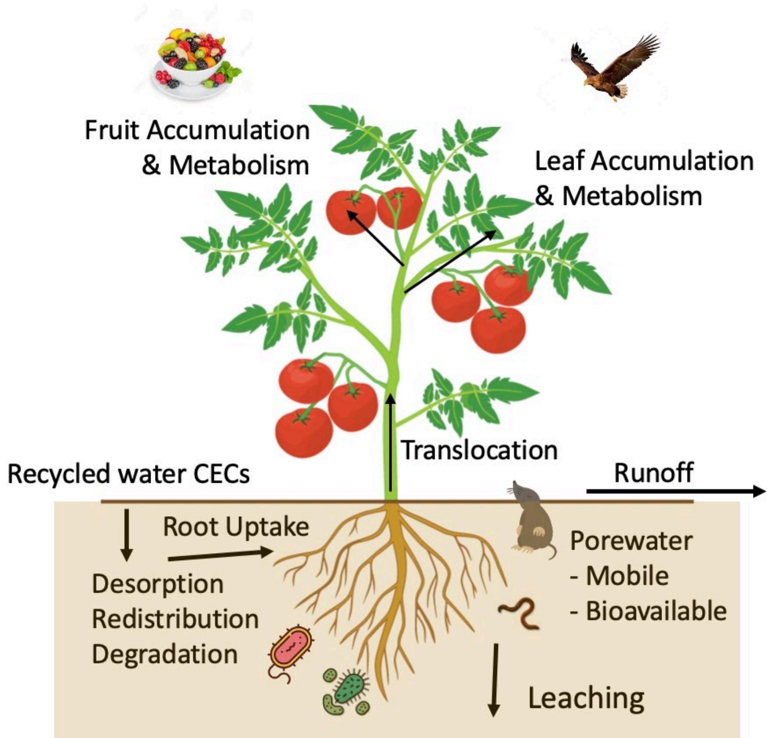

Figure 4-1. Fate Processes of Recycled Water CECs in the Soil-Plant Continuum.
Source: Science of the Total Environment 814(2022); by Shi, Q., Y. Xiong, P. Kaur, N. Darlucio Sy, J. Gan; "Contaminants of Emerging Concerns in Recycled Water: Fate and Risks in Agroecosystems"; p. 152527; Copyright (2022), with permission from Elsevier.

4.2.2.1 CECs in Soil Receiving Recycled Water

A great number of CECs and related metabolites are present in recycled water, with composition and concentrations varying continuously on a spatiotemporal scale (Diwan et al., 2013; Petrie et al., 2015). Some CECs have been found to accumulate in soil, as evidenced from their levels in soil being significantly higher than in the irrigation water (Kinney et al., 2006a; Calderón-Preciado et al., 2011b). For example, the total concentration of tetracyclines was reported in the range of 12.7 to 145.2 µg/g in public parks where long-term recycled water irrigation was practiced (Wang et al., 2014). Many studies reported similar antibiotics residual concentrations in the range of µg/g to mg/g following repeated applications of recycled water for irrigation (Hamscher et al., 2002; Wang and Han, 2008; Zhang et al., 2008a; Zhang et al., 2008c; Zhang et al., 2008b; Li et al., 2009; Tai et al., 2010; Hu et al., 2010; Li et al., 2011). The levels of BFRs, organochlorine pesticides, and polychlorinated biphenyls were surveyed in the surface soils of a vegetable farm in a BFR-manufacturing region in North China, and the total concentrations of BFRs ranged from 0.04 to 8.15 µg/g (Zhu et al., 2014). PFASs in soil were generally low, with levels up to 0.2 µg/g for PFOA (Blaine et al., 2014a). In a study involving river water irrigation in Jordan, ∑PFASs concentration was 3.4×10^{-4} µg/g in an alfalfa soil, 7.1×10^{-4} µg/g in a mint soil and 9.1×10^{-4} µg/g in a lettuce soil, suggesting relatively low accumulation (Shigei et al., 2020).

4.2.2.2 Sorption and Degradation of CECs in Soil

When recycled water is applied to agricultural fields, CECs in the recycled water undergo several processes in the soil-plant system, which affects their accumulation in plants, downward leaching, or surface runoff. These processes often serve the role to limit the chemical flux towards the final receptor. Due to the wide structural diversity of CECs, understanding the different sorption mechanisms in soils is critical for predicting their mobility and availability.

Distribution between soil solid particles and water, or sorption, of CECs generally dictates their availability, and chemicals with strong hydrophobicity or positive charge tend to show strong sorption and hence reduced availability for biodegradation, offsite movement or plant accumulation (Wu et al., 2013; Carter et al., 2014; Fu et al., 2016a; Fu et al., 2016b; Dong et al., 2019). It has been shown that hydrophobic CECs such as fragrances are more recalcitrant to biodegradation, resulting in accumulation in soil (Kalavrouziotis, 2017). Carbamazepine, a relatively persistent chemical with weak sorption, was detected in rain-fed wheat grown in soils previously irrigated with recycled water (Ben Mordechay et al., 2018). Non-polar molecules with low sorption capacity and relatively high solubility are preferentially distributed in soil porewater and have been found to be comparatively easy for plant uptake or offsite transport (Hamscher et al., 2002; Pedersen et al., 2005; Stoob et al., 2007; Wei et al., 2014; Prosser et al. 2015; Fairbairn et al., 2016; Jaffrézic et al., 2017).

Sorption of a chemical in soil is influenced by physicochemical properties of CECs and soil (Li et al., 2013; Dodgen et al., 2014; Kodešová et al., 2015; Wen et al. 2016). Specifically, soil variables such as pH, ionic strength, and organic matter can alter the surface complexation and electrostatic processes by changing the surface charge and sorption sites. CECs can be nonionic, cationic, anionic, and zwitterionic compounds,[103] and their chemical species in a soil depend closely on their pK_a and soil solution pH value (Schaffer and Licha, 2015). Sorption of ionic molecules is controlled by electrostatic interactions, cation exchange, cationic bridging, complexation, and hydrogen bonding,[104,105] while hydrophobic interactions, electron donor-acceptor interactions, and weak van der Waals forces are mechanisms driving the sorption of nonionic compounds (Zhang et al., 2010; Klement et al., 2018; Call et al., 2019; Xu et al., 2021). For example, sorption mechanism of nalidixic acid changed into hydrophobic interactions when the pH was lower than its pK_a, because nalidixic acid exists as a zwitterion (Wu et al., 2018).

Degradation, including abiotic and biotic transformations, is a key process involved in the fate of CECs in agroecosystems, which depends closely on the physicochemical properties of specific CECs, and soil microbial communities, pH, moisture content, and other factors (Monteiro et al., 2009; Davis et al., 2015; Dodgen et al., 2015; Pullagurala et al., 2018; Fu et al., 2019; Jiao et al., 2020). Photodegradation of CECs is considered an appreciable abiotic transformation process that may occur in aqueous and soil environments. Significant photodegradation of various tetracyclines and sulfonamides in water and on soil surfaces was observed experimentally (Thiele-Brun and Peters, 2007). Many CECs exhibit enhanced microbial degradation under aerobic conditions, while they are relatively more persistent under anaerobic conditions. For example, under aerobic conditions, the estimated half-life of bisphenol A in soil and sediments was 3—37 d, while 70-d anaerobic soil or 120-d anoxic estuarine sediment while incubation experiments showed no appreciable degradation (Fent et al. 2006; Flint et al., 2012; Yu et al., 2013; Chang et al., 2014; Yang et al., 2014). Generally, basic compounds with positive or neutral charges are readily degraded in soil, while acidic and anionic compounds are more stable (Siemens et al., 2008). In soil with plants, it is well known that microbial degradation occurs mainly in the rhizosphere where plant root exudates are released, and microbial activity is significantly enhanced compared to the bulk soil. As a convenient source of carbon and energy, root exudates accelerate microbial biodegradation by increasing microbial activity and bioavailability (Miya et al., 2001; Gao et al., 2010; Sun et al., 2013; LeFevre et al., 2013). Antimicrobial agents (e.g., triclosan, triclocarban) and microplastics introduced by irrigation have the potential to alter soil microbial communities, phase distribution, and mobility of CECs (Borgman and Chefetz, 2013; Fu et al., 2019; Conley et al., 2019). Populations and activities of microorganisms in solid matrix, which vary with season and space, combined with physicochemical characteristics of soils, lead to different degradation rates of the same compounds in different soils (Xu et al., 2009a). In some cases, soil organic matter may reduce the bioavailability of CECs serve as an alternative nutrition source for microorganisms,129 and thus inhibit the biodegradation of certain organic compounds (Johnson and Sims, 1993; Alvey and Crowley, 1995; Xu et al., 2009b).

4.2.2.3 Offsite Transport of CECs in Soil

While CECs may persist in soil following irrigation at agronomic rates, subsequent transport of CECs in dissolved or adsorbed form to surface water has not been a research focus to date. In one study, some CECs, such as PPCPs, BFRs and plasticizers were targeted for analysis in surface runoff samples collected from effluent-irrigated fields. However, when detected, many of the CECs were present at concentrations lower than the method detection limits, with the exception of gemfibrozil (190-790 ng/L), carbamazepine (320-440 ng/L), carisoprodol (680 ng/L) and butylbenzenesulfonamide (350-590 ng/L) (Pedersen et al., 2005). For the CECs found above the detection limits, the concentrations were below the known aquatic toxicology threshold values; however, their potential to elicit more subtle effects in aquatic organisms cannot be excluded. In the absence of experimental observations, modeling-based predictions are urgently needed to identify CECs that may have the greatest potential for such offsite transport, and actual runoff movement of such chemicals should then be evaluated experimentally under field conditions.

Given the generally low concentrations of CECs in recycled water and that sorption and degradation work in concert to further attenuate CECs, it may be expected that leaching of CECs through soil profiles would be very limited in most cases. For CECs with appreciable sorption, once they come into contact with the soil surface following irrigation of recycled water, they are likely to remain in the upper soil layers. For example, in Xu et al. (2009c) turfgrass fields were irrigated with recycled water for four months, while ibuprofen, naproxen, triclosan, bisphenol A, clofibric acid, and estrone were detected in the 0-30 cm soil layer, no compound was detected in the leachate collected at 89 cm below the surface. Similar leaching experiments concluded that PPCPs such as carbamazepine and triclosan were retained within the top 40 cm of soil (Walker et al., 2012; Chen et al., 2013; Durán-Álvarez et al., 2015). A 10-year recycled water irrigation study showed that CECs in the soil at the 90-cm depth varied from less than 1 ng/g to 140 ng/g while the concentrations in the drainage water at the same depth varied from non-detectable to the µg/L level (Chen et al., 2013). Analysis of leachate water collected at 90 cm below the surface of turfgrass plots irrigated with recycled water for 6 months showed that only 5 compounds were detected in the leachate at trace levels, even though 14 PPCPs were present in the source water at levels between 1 and 1255 ng/L (Bondarenko et al., 2012). The downward movement of CECs could be significant for those chemicals that are relatively persistent and mobile. For example, the monitoring of PPCPs in Slovenia and Croatia suggested that concentrations of PPCPs in soil and groundwater samples were below detection to 319 ng/g and below detection to 745 µg/L, respectively, and some PPCPs, such as diclofenac, mefenamic acid, caffeine, were detected in soil samples collected at the 150 cm depth, indicating that these chemicals underwent significant downward movement, likely due to heavy rain episodes (Biel-Maeso et al., 2018). Studies have shown that the levels of both perfluorooctane sulfonate and perfluorooctanoate generally increased with the soil depth, likely due to their relatively high solubility in water and limited interaction with aquifer solids (Xiao et al., 2015). The presence of CECs in the groundwater may lead to drinking water contamination and ecotoxicological risks. Due to the low microbial activity in the subsurface soil layers, some CECs may be persistent in the aquifer over a long time and may, therefore, pose a long-term risk for communities relying on the use of groundwater as the primary drinking water source (Oppel et al., 2004).

Soil properties strongly affect the downward movement of CECs, with soil organic matter content being recognized as a dominant factor on the vertical transport of organic chemicals within a soil profile. Walker et al. (2012) investigated the concentrations of carbamazepine in soil under different types of land uses, following treated wastewater irrigation for more than 25 years. Forest surface soil (at a depth of 1 cm) exhibited much higher levels of carbamazepine (4.92 ng/g) as compared to cropped areas (1.98 ng/g) at the same depth, likely resulting from the significantly higher organic carbon content in the forest areas. Additionally, most of the carbamazepine was accumulated in the top 30 cm of the soil profile, and little was detected below the 30-cm depth where the soil started transitioning into higher clay and lower organic carbon (Walker et al., 2012). The increment of soil organic carbon, such as the application of compost, could retain some of the CECs near the surface and thus reduce its leaching to groundwater (Filipović et al., 2020). However, prevalence of a large number of synthetic organic compounds in biosolids may pose a nother potential risk for CECs when biosolids are land applied (Wu et al., 2015).

Most studies to date have been limited to discrete subsets of CECs under different experimental conditions, making it difficult to draw general conclusions. Moreover, many studies are also descriptive in nature, reporting the presence or absence of the targeted CECs. Therefore, underlining mechanisms such as chemical mobility or bioavailability, and runoff and leaching following long-term applications of recycled water irrigation, need to be better evaluated under representative conditions.

4.2.3 Plant Uptake, Metabolism and Accumulation of CECs

4.2.3.1 Plant Uptake and Translocation

When recycled water is used to irrigate plants in agricultural fields, CECs may contaminate plants after uptake via roots or leaves. Under the assumption that the majority of irrigation water is received by the soil, roots should serve as the primary route for plant uptake of CECs. The uptake of CECs by plant roots are mainly through passive, diffusive processes, although active uptake through transporters is likely for some compounds, as shown for hormone-like compounds like the phenoxy acid herbicides (Bromilow and Chamberlain, 1995). From hydroponic studies, the root uptake of nonionized chemicals consists of two components: (1) “equilibration” of the aqueous phase in the plant root with the surrounding solution, and (2) “sorption” of the chemical onto lipophilic root solids (Collins et al., 2006). To date, more than 100 CECs have been considered in the evaluation of uptake by plants, including vegetables and fruit-bearing plants (Table 3-1 in Appendix) (Wu et al., 2015; Miller et al., 2016; Bartrons and Peñuelas, 2017). Plant uptake and accumulation of CECs are often measured by bioconcentration factor (BCF), which is calculated as the ratio of the concentration of a chemical in the plant tissue to that in the soil porewater. Most published studies have shown detectable levels of CECs in plant roots in both hydroponic and soil settings (Wu et al., 2015; Miller et al., 2016). BCF values of CECs in roots, also known as root concentration factor (RCF), varied widely between hydroponic and soil cultivations. RCFs of CECs have been found to reach up to 840 L/kg in hydroponic experiments, while the values obtained from soil experiments were much smaller for the same compounds, ranging from background to 40 L/kg, suggesting that the uptake of CECs by plant roots in soil largely depends on their bioavailability in soil porewater in the rhizosphere (Wu et al., 2015). Chemicals with strong sorption to soil are expected to be present predominantly in the sorbed form and therefore are less likely to be accumulated into plant roots.

Studies have been performed under laboratory or greenhouse conditions to understand the bioaccumulation potentials and uptake mechanisms of different classes of CECs in plants. Some pharmaceuticals have been found to be more easily absorbed by plants. As a commonly used medication in the world, carbamazepine has been often detected in various plants irrigated with recycled water. This is likely due to the fact that carbamazepine has a relatively high bioavailability in soil with weak adsorption and long persistence (Wu et al., 2010; Shenker et al., 2011; Goldstein et al., 2014). Significant uptake was observed for the antidiabetic metformin in carrot and barley plants (RCF ~8), as compared with the antibiotics ciprofloxacin and narasin (RCF ∼2) in a sandy soil spiked with target chemicals (Eggen et al., 2011) . In contrast, only less than 0.1% of the ibuprofen added with spiked water was transferred to the roots of ryegrass (Winker et al., 2010).

The mechanism for the translocation of specific CECs varies with the physicochemical properties of CECs. For example, factors influencing uptake and translocation of organic pollutants are summarized in Table 4-1. Once taken up, CECs reaching the vascular tissue can be transported to shoots, leaves, and fruit via the xylem or phloem (Kvesitadze et al., 2015). The movement of organic contaminants in the xylem sap is controlled by proteins transporter while in phloem it is predominantly driven by osmotic pressure gradient (Satoh, 2006; Turgeon and Wolf, 2009; Inui et al., 2013). The translocation of hydrophilic CECs are more favorable in the Casparian strip as compared to hydrophobic compounds. Indeed, greenhouse experiments showed that the accumulation of caffeine in xylem sap was greater than those of triclocarban and endosulfan in soybean, zucchini and squash plants (Garvin et al., 2015). Similarly, Tanoue et al. (2012) demonstrated that compounds with an intermediate polarity, such as carbamazepine and crotamiton, were readily transported from cucumber roots to shoots under hydroponic conditions.

Numerical models have been used for predicting plant uptake of chemicals, ranging from empirical to mechanistic models. Empirical models were proposed and applied for correlating the concentration of chemicals in plant tissues with its physicochemical properties (Topp et al., 1986; Travis and Arms, 1988). For example, Hyland et al. (2015) linked BCF, and transpiration stream concentration factor (TSCF) with physicochemical properties, such as pH-adjusted octanol-water partition coefficient (D_{ow}) and reported a positive correlation between log RCF and log D_{ow} for 9 pharmaceuticals and flame retardants in roots of lettuce and strawberry grown in soil ($R^2 = 0.78$). The model developed by Hwang et al. (2017) demonstrated a high correlation between modeled and measured concentrations with R^2 values of 0.97 to 0.98 by taking chemical migration and dissipation in soil, plant transpiration stream, root–soil transfer rate, and plant growth into consideration. However, due to their theoretical limitations, empirical models only provide a coarse description of the plant uptake processes. A more comprehensive study with acids, bases, and neutral pharmaceuticals showed that the correlation was strong only when neutral compounds were considered (Miller et al., 2016).

A more sophisticated approach is the use of mechanistic multicompartment dynamic plant uptake (DPU) models derived from a classic mechanistic approach (Hung and Mackay, 1997). All major physicochemical processes happening during the DPU process are encompassed in this model (Rein et al., 2011; Trapp, 2007). Both neutral compounds and ionic chemicals were tested successfully against measured data under different operating conditions (Trapp, 2009). Furthermore, Prosser et al. (2014) applied the DPU model and biosolids-amended soil level IV model to predict concentrations of 8 PPCPs in the tissue of plants grown in a biosolids amended soil. Additionally, a widely used hydrological model (HYDRUS) was coupled with a multicompartment DPU model to account for differentiated multiple metabolism pathways in plant tissues.

Table 4-1. Factors Influencing Uptake and Translocation of Organic Pollutants by Plants.

Factors		Potential Uptake by Plants: +	Potential Uptake by Plants: -	Comments	References
Organic contaminants	Molecular mass	Low molecular mass	High molecular mass	Usually <500 Da and volatile compounds	Limmer et al., (2014)
	Hydrophobicity	Moderately hydrophobic and a few hydrophilic	The rest	Not useful factor for ionic compounds uptake.	
	Polarity	High polarity	Low Polarity	For neutral organic compounds, sorption in soil tends to decrease as compound polarity increases. So, compounds are more accessible for uptake.	Miller et al., (2016)
	K_{OW} (Octanol-water partition coefficient)	High K_{OW}	Low K_{OW}	Most translocatable compounds: Log K_{OW} between 1-4. Usually absorbed by plant roots.	Limmer et al., (2014) Zhang et al., (2017)
	K_{OA} (Octanol-air partition coefficient)	Low K_{OA}	High K_{OA}	Only for semi-volatile or volatile compounds. Absorbed by plants from air.	Zhang et al., (2017)
Plant biological characteristics	Root extractable lipid content	Low lipid content in roots	High lipid content in roots	Positive correlation between organic contaminants content and lipid content in plant roots.	Gao et al., (2005)
	Genotype	Leafy vegetables	Succulent plants		
	Healthy	Non-stressed plants	Stressed plants		
	Irrigation requirements	High net irrigation requirements	Low net irrigation requirements		
	Transpiration stream concentration factor (TSCF)	High plant evapotranspiration	Low plant evapotranspiration	The TSCF can show the capacity of organic pollutant translocation from roots to aboveground parts. Maximum values of TSCF for chemicals with log $K_{OW} \approx 1.8$	Collins et al., (2006)
	Permeability of biomembranes	High permeability	Low permeability	Membrane characteristics can be used to predict accumulation concentrations of organic pollutants.	Sterling et al., (1994)
Soil properties	Aeration	Aerated soils (aerobic conditions)	Waterlogged soils (anaerobic conditions)		Goldstein et al., (2014)
	SOM (Soil Organic Matter)	Sandy soil	Clay/Loamy soils		Goldstein et al., (2014)
	Humidity	Adequate soil moisture	Drought		
Environmental media	Humidity	Low air humidity	High air humidity		
	pH of environmental media	Acidic pH (pH<pKa of organics)	Basic pH (pH>pKa of organics)		
	Temperature	Hot and dry agricultural areas	Cold agricultural areas	Higher temperature coefficient for diffusion processes of organic pollutants can accelerate passive absorption by the plant.	
	Wind speed	High wind speed	Calm/low wind speed		

4.2.3.2 Influencing Factors

The physicochemical properties (K_{ow}, pK_a, charge state, and molecular weight) of CECs significantly affect their uptake and translocation processes in the soil-plant system. For non-charged organic compounds octanol–water partition coefficient (log K_{ow}) was suggested as a predictor for uptake behavior (Hsu et al., 1990; Sicbaldi et al., 1997; Dettenmaier et al., 2009). Nonionized hydrophilic compounds (log K_{ow} < 0) are mobile in both xylem and phloem whereas the compounds of intermediate lipophilicity (0 < log K_{ow} < 3) are only xylem-mobile due to their high affinity to lipidic membranes (Kalavrouziotis, 2017). Generally, compounds of intermediate hydrophobicity have the highest translocation through the plant compartments, as compared to compounds outside this range (Briggs et al., 1982). Furthermore, to calibrate for the effect of pH on ionization, acid-base coefficient (pK_a) and medium pH are used to adjust K_{ow} to the pH-adjusted D_{ow} (Eq. (4-1)) (Hsu et al., 1990; Hyland et al., 2015). The translocation from soil to root was found to correlate positively with log D_{ow}, driven by chemical sorption to the root surfaces, while the transport from roots to leaves was negatively related to log D_{ow}, suggesting hydrophilicity-regulated transport via xylem (Wu et al., 2013; Hyland et al., 2015). Besides partitioning coefficients, molecular weight was found to be another factor affecting chemical uptake and translocation. Compounds with lower molecular weights are more prone to be taken up by plants. For example, short-chain PFASs were found to readily transport and bioaccumulate in edible tissues, likely because smaller molecules are easier to cross the Casparian strip (Suga et al., 2003; Müller et al., 2016). Similar correlations were previously found for PAHs, where 2–4 ring PAHs were detected in wheat grown in soil than 5 or 6-ring PAHs (Li and Ma, 2016).

$$\log D_{OW}(pH) = \log K_{OW} - \log(1 + 10^{(pH - pK_a)\Delta i}) \qquad \text{(Equation 4-1)}$$

where: $\Delta i = 1$ *for acids and -1 for bases*

Soil properties, including pH, soil organic matter content and texture, infiltration rate, nutritional conditions, exposure concentrations and duration, as well as metabolic inhibitors were reportedly involved in regulating plant uptake of CECs (Khan et al., 2008; Xu et al., 2009a; Trapp, 2007; You et al., 2010; Wen et al., 2013; Zhao et al., 2013; Wen et al., 2015; Zhan et al., 2015; Guo et al., 2016; Xing et al., 2016). The specific surface areas and composition of soil are some of the most relevant factors affecting the bioavailability of CECs in rhizospheres. For example, organic matter content was found to be positively correlated with the sorption of neutral CECs. Therefore, in soils with high organic matter and clay contents, plant uptake of nonionic and some ionizable CECs, such as carbamazepine, sulfapyridine, lamotrigine and caffeine, was reduced, likely due to decreased chemical concentrations in the soil porewater (Tolls, 2001; Shenker et al., 2011; Goldstein et al., 2014).

The co-existence of other contaminants and biosolids amendment may further influence the bioavailability of CECs for plants. Biosolids generally contain 40-85% organic matter on a dry mass basis (Kinney et al., 2006b), and consequently the presence of biosolids may greatly decrease the mobility and plant uptake of CECs. For example, the bioavailability of PPCPs, PFAS, and polybrominated diphenyl ethers (PBDEs) was reduced after addition of biosolids and biochars due to increased sorption of CECs (Caicedo et al., 2011; Gellrich et al., 2012; Fu et al., 2016b; Jachero et al., 2016; Zhu et al., 2018), while the introduction of plant alkaloids into the soil increased the uptake of the PAHs (Navarro et al., 2009).

4.2.3.3 Plant Metabolism

After entering the root, CECs may be metabolized in the plant (Miller et al., 2016; Hurtado et al., 2016). Most CECs undergo a sequential metabolism in plants and are transformed into more hydrophilic and less toxic compounds, which decreases the ultimate potential accumulation in edible tissues (Fu et al., 2017a; Fu et al., 2017b; Fu et al., 2018; Dudley et al., 2018; Sun et al., 2019). Similar to the detoxification processes in an animal liver, enzyme-assisted chemical transformations take place in plants and this process in plants is known as “green livers”(Sandermann, 1992). In plants, the detoxification of xenobiotics generally occurs in three consecutive phases, as illustrated in Figure 4-2 (Coleman et al., 1997). During phase I, xenobiotics are transformed into more polar or reactive intermediates through hydroxylation, hydrolysis or other redox-based reactions. In subsequent phase II, glutathione, sugars, or amino acid-assisted conjugation of metabolites takes place. In phase III, the inert conjugated xenobiotics are either stored in vacuoles or incorporated into cell walls (Sandermann, 1992; Ohkawa et al., 1999; Dietz and Schnoor, 2001).

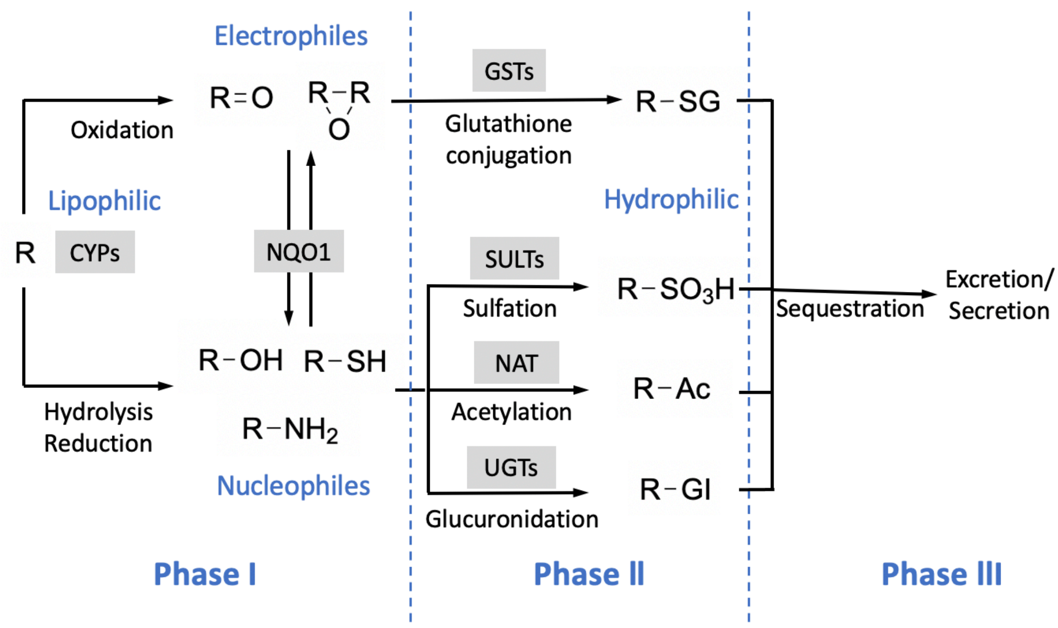

Figure 4-2. Major Metabolism Pathway of CECs in Plants. Abbreviated Xenobiotic-Metabolizing Enzymes: CYPs, Cytochromes P450; NQO1, NAD(P)H:quinone Oxidoreductase 1; UGTs, UDP-glucuronosyltransferases; SULTs, Sulfotransferases; NAT, N-acetyltransferases; GSTs, Glutathione S-transferase.

At present, in-plant processes, metabolic reactions, and transformation products of CECs are largely unknown, as metabolism is strongly specific to the chemicals as well as plant species. Generally, conjugation with biomolecules is the key process in the detoxification of CECs by plants. Transformed metabolites are combined with natural molecules like sugars, amino acids and malonate, catalyzed by enzymes such as glycosyltransferase, glutathione *S*-transferases, peroxidases, and hydrolases, and consequently increase the hydrophilicity of the parent compounds (Wu et al., 2021). For example, naproxen, diclofenac, ibuprofen, and gemfibrozil are likely to form glycoside conjugates in phase II metabolism in the carrot cell culture (Wu et al., 2016). However, bioactivation could be a potential concern with some CECs (Malchi et al., 2014; Goldstein et al., 2014). For example, the metabolite of carbamazepine, 10,11-epoxycarbamazepine, was shown to be more toxic than the parent compound (Malchi et al., 2014). Furthermore, direct conjugation may mask the parent compound or its bioactive metabolites by preserving the bioactivity (LeFevre et al., 2015; Fu et al., 2017a; Fu et al., 2017b; Huynh et al., 2018). Once the plant materials are ingested, for instance, the chemicals may be released through deconjugation and back conversions (Sakamoto et al., 2002; Claus et al., 2016).

Studies on plant metabolism of some CECs have shown that plants are capable of taking up precursors and transform them to the more stable terminal metabolites. A previous study showed that >81% *N*-ethyl perfluorooctane sulfonamide was transformed into intermediate metabolites perfluorooctane sulfonamido acetate and perfluorooctane sulfonamide, as well as the terminal metabolite perfluorooctane sulfonate (PFOS) in a 182-d soil-carrot (*Daucus carota ssp sativus*) mesocosm experiment (Zabaleta et al., 2018). The apparent half-life of N-ethyl perfluorooctane sulfonamide was 13.9 ± 2.1 d in the aerobic soil, but if considering transformation products, the persistence was substantially prolonged (Mejia Avendaño and Liu, 2015).

In lieu of whole plants, plant callus tissues or cell cultures have been used as a model system due to their small biomass, rapid growth, and the need for only small amounts of chemicals. Plant cells have been used as model systems to investigate the metabolism of various xenobiotics, such as 4-nonylphenol, bisphenol A, phthalate esters, and PPCPs (Bokern et al., 1996; Schmidt and Schuphan, 2002; Sun et al., 2015; Wu et al., 2016; Fu et al., 2017a; Fu et al., 2017b; Dudley et al., 2018; Dudley et al., 2019; Cheng et al., 2020). Wu et al. (2016) used carrot cell cultures as a rapid screening tool to assess the metabolism of 18 PPCPS in 96-h incubation experiments and found that 7 PPCPs, including triclosan, naproxen, diclofenac, ibuprofen, gemfibrozil, sulfamethoxazole, and atorvastatin, displayed rapid metabolism, with the estimated 50% dissipation time ranging from 0.8 to 17 h. No appreciable metabolism was observed for the other 11 PPCPs, likely due to their resistance to metabolism. Previous studies have shown that similar metabolites and metabolism pathways occur in plant cell cultures and intact plants (Kolb and Harms, 2000). Therefore, the use of plant cell cultures, instead of the whole plants, could be a promising tool for rapid screening of metabolism potentials of CECs and identifying CECs that have a high likelihood for plant accumulation due to recalcitrance to metabolism.

4.2.3.4 Phytotoxicity

Phytotoxicity is the effect of chemicals on plants causing external physiological stresses. A number of studies have shown that CECs can induce toxicity in plants and terrestrial organisms (Wu et al., 2013; Prosser et al., 2015; Sun et al., 2018). Even at low levels, chronic exposure may compromise the fitness of plants and tip hormone balances (McGinnis et al., 2019). Physiological parameters including seed germination and root elongation have been used to evaluate the phytotoxicity effects. For example, when the level of hexabromocyclododecane (HBCD) reached 0.05 mg/L in a hydroponically grown maize system, the inhibition rate was

46.54, 31.94 and 11.97% for germination rate, root elongation, and shoot elongation, respectively, after 4-d exposure (Wu et al., 2017). Similar results were observed for antibiotics like chlortetracycline, enrofloxacin, and sulphathiazole in laboratory soil experiments (Chung et al., 2017). It is worth noting that PFASs slightly stimulated wheat seedling growth at low levels (< 10 mg/L) while exerted a significant inhibitory effect on root elongation when the level exceeded 200 mg/L in hydroponic experiments (Qu et al., 2010).

In addition to the physiological evaluation, molecular level changes were studied to understand the mechanism of phytotoxicity. When the concentration of PFOS reached 200 mg/L, the rate of superoxide dismutase and peroxidase activity was inhabited by 12.6% in roots and 27.9% in leaves, which further led to overproduction of reactive oxygen species (ROS) (Qu et al., 2010). As ROS levels increased, DNA damage and protein denaturation led to the death of plant cells (Wu et al., 2016). Likewise, the activity of guaiacol peroxidase was modified and the profile of proteins was changed after the uptake of tetracycline in pea seedlings (Margas et al., 2016). In addition to the effects on growth and development, the metabolites of some antibiotics were found to be more harmful than the parent compounds, such as the genotoxic hydrazine-containing metabolites in spring onion transformed from nitrofuran (Di Marco et al., 2014).

Recycled water-borne CECs are applied to agricultural fields in a mixture, but only a few studies have considered the cocktail effects of CECs to plants (Fu et al., 2019). Exacerbated stress-related effects, including membrane lipid peroxidation and oxidative burst, were observed in roots of alfalfa treated with the mixture of diclofenac, sulfamethoxazole, trimethoprim and 17α-ethinylestradiol. Even though potential effects to plants elicited by individual CECs were generally low (Migliore et al., 2003), serious oxidative stress was reported in cucumber plants exposed hydroponically to a mixture of 17 PPCPs at an environmentally relevant dose (50 µg/L) (Sun et al., 2018).

It is important to note that many of the studies were conducted under hydroponic conditions with chemical concentration well above environmentally relevant levels. The actual phytotoxicity from recycled water irrigation due to CECs could be much less significant. Therefore, further studies are needed to evaluate physiological and biological responses of plants under field conditions, and to differentiate effects from CECs and other constituents such as nutrients and salts in recycled water.

4.2.4 Risks to Human and Environmental Health

Some CECs, such as many pharmaceuticals, are designed to have biological activity at low concentrations, while many others (e.g., bisphenols, phthalates, flame retardants) may have unintended adverse effects, such as endocrine-disrupting activity and developmental toxicity (Diamond et al. 2015). To promote the safe reuse of treated wastewater in agriculture, it is crucial to understand the human health risks imposed by CECs, such as human exposure through dietary intakes of contaminated food produce. Currently, there is no comprehensive evaluation of human dietary exposure of CECs. Only a few studies have attempted to predict the probable exposure of some CECs. Findings to date suggest that CECs in food crops would not be considered potentially hazardous for daily consumption, since the quantities of CECs taken in by humans through consumption would be well below human exposure limits (Wu et al., 2013; Wu et al., 2014; Wu et al., 2015; Mendez et al., 2016; King County 2019). Two approaches are commonly used to assess the human exposure risk: (1) acceptable daily intake (ADI) calculated by applying 100-fold safety or uncertainty factor to the no observed adverse effect level obtained from the most sensitive experimental tests (Fuquay et al., 2011), and (2) the threshold of toxicological concern (TTC) concept based on Cramer classification, which is used to estimate the exposure threshold values for chemicals with known structures (Cramer et al., 1976; Kroes et al., 2004). Prosser and Sibley (2015) assessed risk by comparing estimated daily intake values with ADI and concluded that most individual CECs in plants introduced by wastewater irrigation would not pose a hazard to human health. In a field-grown vegetable study, Wu et al. (2014) estimated annual exposure of 9 pharmaceuticals through the consumption of 8 vegetables irrigated with fortified recycled water, and the values 3.69 μg per capita, which was only about 0.003% of the minimum single medical dose for therapeutical treatment (10−200 mg/day for adults). Similar results were reported by Riemenschneider et al. (2016) with 0.003−15 ng/kg daily exposure of body weight for nine pollutants, including PPCPs and carbamazepine metabolites. The same study also listed the limits of daily vegetable consumption for adults to reach the respective TTC, and the values were more than 9 kg for pharmaceuticals and their metabolites, except for 10,11-epoxycarbamazepine and ciprofloxazin, with daily consumption of less than 0.55 kg for all the vegetables, suggesting the need for consideration of specific toxicological characteristics of these chemicals (Riemenschneider et al., 2016).

To date, most human health risk assessments have been based on theoretical calculations, with uptake experiments conducted under controlled conditions, while the actual exposure from commercially available produce is essentially unknown. In a randomized controlled trial, healthy individuals were offered commercially available produce grown in soils irrigated with recycled water (Paltiel et al., 2016). The trial showed that carbamazepine and its metabolites were detected in urine samples, while the levels were undetectable or significantly lower in subjects consuming fresh water-irrigated produce, suggesting *in-vivo* exposure of humans to CECs via consumption of crops irrigated with recycled water (Paltiel et al., 2016).

Application of recycled water to agricultural fields also brings CECs into contact with soil-dwelling organisms including microorganisms and earthworms that play important roles in maintaining soil health and sustaining various terrestrial food chains. Soil phosphatase activity was found to be significantly inhibited in a loamy soil 22 d after the treatment of six antibiotics (chlortetracycline, tetracycline, tylosin, sulfamethoxazole, sulfamethazine and trimethoprim) at concentrations between 1 and 300 mg/kg (Liu et al., 2009). Waller and Kookana (2009) showed that triclosan at 1 mg/kg or above was capable of disturbing the soil nitrogen cycle. Since some microbes and plants share symbiotic relationships, disturbance in microbial community structures may further perturb the equilibrium of soil ecosystems. Several studies have considered adverse effects of some CECs on earthworms. In agricultural soils from Beijing and Tianjin suburbs receiving recycled water and biosolids, the concentration of tetracyclines was found up to 119–307 mg/kg (Xie et al., 2011a). Based on the environmental concentrations, Lin et al. (2012) found that exposure to chlortetracycline at 100 mg/kg in an agriculture soil for 28 d led to the reduction of 45.6% and 43.2% in the number of cocoon and juvenile earthworm (*Eisenia fetida*), respectively, while the reduction further increased to 63.9% and 68.6% at 300 mg/kg. Although no effect was observed on the mortality of adult worms, DNA damage in coelomocytes was observed to increase to 48% in a soil containing 30 mg/kg chlortetracycline (Lin et al., 2012). Similarly, DNA damage and enzyme activity in earthworm were reported with the exposure to triclosan (Kinney et al., 2008), triclocarban (Snyder et al., 2011), caffeine (McKelvie et al. 2011), decabromodiphenyl ether (Xie et al., 2011b), and decabromodiphenyl ethane, (Hardy et al., 2011) indicating the potential genotoxicity of some CECs on earthworms.

In addition, insects may also be impacted by CECs accumulated in plants. Pennington et al. (2017) observed an increase in both developmental time and mortality in cabbage loopers (*Trichoplusia ni; Hübner, Lepidoptera: Noctuidae*) when they were reared on plant tissues containing antibiotics (lincomycin, oxytetracycline, and ciprofloxacin), hormones (estrone, 19-norethindrone, 17β-estradiol, and 17α-ethynylestradiol), or a mixture of such CECs. The ecological impacts of these behavioral and metabolic changes are unknown under field-relevant conditions. Additionally, studies have so far identified additive and synergistic effects of mixtures of CECs on aquatic organisms (Flaherty and Dodson, 2005), however research into soil invertebrates is lacking (Pedersen et al., 2005).

Based on the concentrations observed in the field, acute effects such as lethality of nontarget organisms seem to be unlikely; however, subtler effects may not be excluded, which, over long time, may lead to ecological consequences. The general lack of overt acute toxicity data, combined with the challenge in assessing subtle physiological and behavioral changes in exposed organisms, limits a more accurate assessment of toxicological risk of CECs (Wiklund et al., 2012). Our knowledge on CECs in terrestrial ecosystems, such as agroecosystems irrigated with recycled, is still far too limited to draw any concrete conclusions. Further studies, including field observations with typical uses of recycled water at agronomic irrigation rates, are needed to advance our understanding of both the human health and ecological risks of CECs introduced through the beneficial use of treated recycled water.

4.3 Further Research Directions

Our knowledge about the fate and risks of CECs introduced via the use of recycled water in agroecosystems is still very limited at present. The available scientific literature has not shown any concrete evidence for detrimental effects of CECs on human health as the result of use of recycled water for irrigation. Given the generally trace levels of CECs, it is more probable to anticipate adverse effects on non-target organisms such as soil-dwelling invertebrates and insects. However, it is important to note that CECs encompass numerous chemicals and their metabolites, and that CECs are extremely diverse in structures and physicochemical properties. The sheer number of CECs and their metabolites precludes comprehensive evaluations using experimentation-based approaches. There is a great need to develop a short list of priority CECs that may pose the greatest environmental or human health risks. Future research efforts should then be devoted to focusing on the high priority CECs. Below they outline a few topics that merit urgent attention for research:

1. **Prioritization of CECs for Future Evaluation**
   At present, there are over 40,000 organic compounds that are potential CECs and the number is expected to further increase as our analytical ability continues to improve and

new compounds are introduced into the circulation (Diamond et al., 2011). Given the large number of CECs, it is infeasible to evaluate all CECs through experiments. Thus, there is an urgent need to identify a short list of priority CECs in recycled water with high risk for plant uptake and accumulation in food products and/or potential adverse effects on human health and ecosystem to guide future research and regulation directions. To accomplish this purpose, dynamic models for CEC fate prediction in different environmental compartments under recycled water reuse scenarios as well as model validation with field derived data are vitally needed. Multiple variables, including solubility, hydrophobicity, charge state, soil organisms as well as synergistic or antagonistic effects of CECs mixture, need to be incorporated into models to improve predictions, as independent mechanisms-based models are likely less effective for realistic conditions. Simultaneously, some rapid screening tools, such as the use of plant cell cultures, can be exploited to evaluate plant uptake and metabolism potentials to exclude low priority CECs, thus helping identify high priority CECs.

2. **Evaluation under Field Conditions**
   Most studies to date have been carried out under simulated conditions, such as hydroponic systems, to assess the plant uptake of CECs. Offsite movement, plant uptake and ecotoxicity of CECs in the soil-plant system are highly specific to water quality, irrigation methods and frequency, soil properties, climate zones, planting conditions, among other factors. While hydroponic settings are useful in elucidating influences of physicochemical properties and plant species on accumulation of CECs, many fields relevant factors are omitted, such as sorption-desorption at the soil-water interface, water movement in soil, and rhizosphere microbial activities. Therefore, results from simplified systems such as hydroponic systems may not be readily extrapolated to predict the behavior of CECs in the field. In an actual cultivated field, after CECs enter the soil, they are subject to a multitude of processes working in concert to affect their potential for offsite movement or plant accumulation. Sorption by soil and microbial degradation would imply that only relatively persistent CECs that are weakly sorbed by soil may be transported offsite via leaching or runoff. For CECs in soil porewater, plant roots serve as an effective barrier, allowing appreciable uptake of only certain CECs into a plant. Once inside a plant, translocation, along with extensive metabolism, would serve as additional mechanisms to further minimize the likelihood for CECs to accumulate in edible plant tissues. Therefore, the short-listed CECs should be evaluated under representative field conditions to gain first-hand information that may be used to provide a precautionary estimate of potential human health and ecological risks.
3. **Long Term Eco-Toxicological Effects in Agroecosystems**
   To date, most research on the adverse ecological effects of CECs has been focused on surface aquatic systems, and only a small number of studies have considered terrestrial organisms (Brausch et al., 2012). At the levels reported to date, acute effects such as lethality of nontarget organisms appear to be unlikely, but more subtle effects could potentially result in adverse ecological consequences. More studies considering important terrestrial invertebrates are warranted to identify chemical classes of concern and help ensure the ecosystem service of terrestrial environments impacted by the use of recycled water.
4. **Potential Risks of CEC Mixtures and Metabolites**
   Conventional ecological risk assessment approaches use individual compounds and the

acceptable or minimal risk exposure levels derived from linear extrapolations of high-dose exposures, which creates great uncertainties by neglecting the complexity of CEC mixtures. In addition, biologically active metabolites may be produced during the transformation of CECs in soil and plants (Li et al., 2013; Dodgen et al., 2014), further contributing to the mixture of CECs. Screening and identifying unknown CEC metabolites are particularly challenging (Fu et al., 2019). Therefore, state-of-art nontarget screening methods are needed to obtain a more comprehensive picture of CEC mixtures in the terrestrial ecosystems.

5. **Minimizing Risks by Pairing Recycled Water Irrigation with Crop Types**
   Agricultural crops vary greatly in many aspects, including their physiological characteristics, growing seasons, as well as cultivation practices. It is highly likely that CECs would pose lesser of a risk for human exposure when recycled water is used on crops such as feed crops and fruit trees. Most feedstock crops, e.g., alfalfa, Jordan grass, forage corn, have large biomass, which would result in greater biological dilution for CECs. Based on limited field data, translocation and accumulation of CECs in fruits seem to be rather limited as compared to roots or leaves. Irrigation of recycled water in fruit tree orchards may represent a considerably safer alternative than, e.g., vegetables. Figure 4.3 shows potential for CEC uptake by various plants. Research is urgently needed to compare potential environmental and human risks when recycled water is used for irrigating different agricultural crops, and using different irrigation methods (e.g., sprinkler versus drip irrigation).

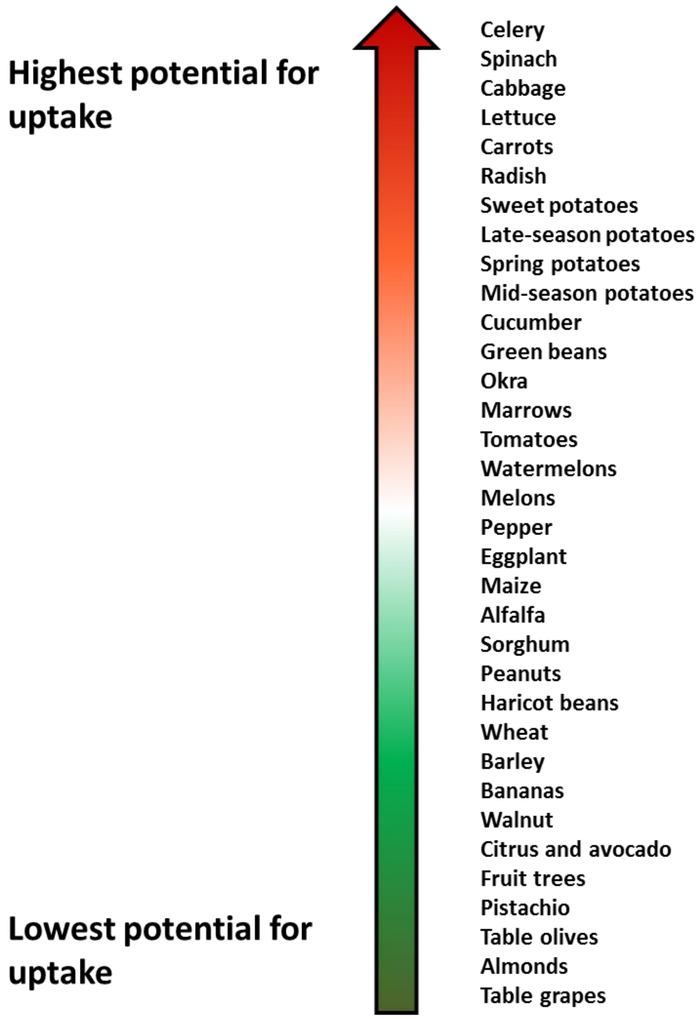

Figure 4-3. Potential Uptake of CECs in Various Crops.

Source: Reprinted from *Environmental Research* 170(2019); by A. Christou, G. Papadavid, P. Dalias, V. Fotopoulos, C. Michael, J. Bayona, B. Piña, and D. Fatta-Kassinos; "Ranking of Crop Plants According to their Potential to Uptake and Accumulate Contaminants of Emerging Concern"; p. 422-432; Copyright (2019), with permission from Elsevier.

4.4 New Trends in Wastewater Treatment for Removal of Contaminants of Emerging Concern

The first approach to lowering the health and environmental risk from wastewater use in agriculture is adequate wastewater treatment. However, the quality of the wastewater and its nature vary enormously, both between and within countries. In many countries in Africa, Asia and Latin America, wastewater tends not to be treated, while in middle-income countries, such as Tunisia and Jordan, and in developed countries, treated wastewater is used. The design of WWTP is crucial to minimize or remove contaminants of emerging concern in effluents used directly or indirectly for crop production. The selection of technologies should be environmentally sustainable and designed to adapt to local conditions. Other factors such as low installation and maintenance costs as well as ease of operation and maintenance are required.

Conventional wastewater treatment consists of a combination of physical, chemical, and biological processes. However, conventional processes are not designed to remove contaminants of emerging concern. So, advanced treatments technologies are required. The new trends in the use of these treatments are based on the interest that developed countries have in CECs. Different methodologies are currently being studied, and the efficiency depending on the wastewater characteristics, the concentration of CECs, their physical-chemical properties and the treatment conditions. Some examples are described in Table 4-2.

In general, oxidation with ozone and adsorption using activated carbon has been successfully tested in large scale applications and usually are the most economical options. However, in case of activated carbon an additional disinfection step is necessary for agricultural reuse practices. In the case of ozonation, potential transformation products generated during the process also need to be considered. Other technologies such as Fenton or photo-Fenton processes have shown very good results in recent years in terms of CECs removal. However, the implementation at full-scale has been limited due to high construction and operation costs. Due to the advantages and disadvantages of all these methodologies, a combination of different processes could be necessary.

Table 4-2. New Trends in Wastewater Treatment Plants (WWTPs) Design for Contaminants of Emerging Concern (CECs) Removal.

Treatment			Advantages	Drawbacks	Recommendation	Full Scale applications
Filtration	Nanofiltration (NF) and Reverse Osmosis (RO)		-High CECs removal -RO can reduce partially effluent salinity -Full rejection of particles -Effective as disinfection process too	-High energy is required -High investment costs -Generation of waste stream (concentrate) containing chemical pollutants -Need of pre-treatment to remove solids	-Complementary treatments for concentrates are needed (usually AOPs). -Cost reduction if combined with other technologies (i.e., MBR).	-RO is usually used in potable reuse treatment
Adsorption	Activated Carbon	Powdered activated carbon (PAC)	-High CECs removal -Additional DOC removal -No by-products are generated -Low initial cost and flexibility of dosage -Adding to a conventional active sludge system improves stability and nitrification.	-PAC must be disposed -A post-treatment for separation of residual PAC is needed (membrane, textile or sand filter). -Production of PAC needs high energy. -Adsorption capacity may fluctuate with each batch.	-Monitoring strategy to control adsorption capacity fluctuations. -For agricultural uses additional disinfection step is needed.	Full scale evidence on practicability in Switzerland and Germany
		Granular activated Carbon (GAC)	-High CECs removal -Additional DOC removal -No by-products are generated -An existing sand filtration can relatively easily be replaced by GAC.	-Production of PAC needs high energy. -No flexibility of dosage to react at changes in wastewater composition. -Adsorption capacity may fluctuate with each batch. -Replace of GAC is required due to a decrease in available adsorption sites.	-Monitoring strategy to control adsorption capacity fluctuations. -Regeneration of GAC. -For agricultural uses additional disinfection step is needed	(2)
Advanced Oxidation Processes	Homogeneous	UV/H_2O_2	-Moderate CECs removal -Use of solar irradiation -Effective as disinfection process too	-Generation of by-products that can be toxic. -Longer reaction time compared with photo-Fenton	-Toxicity tests before effluent discharge	No full-scale evidence on CECs removal
		Fenton (Fe/H_2O_2)		-Typically effective under acidic conditions (pH 3) -Final separation of soluble iron species are required -Addition of complexing agents for working at neutral pH are required	-Additional processes in order to decreased and subsequently neutralized pH values before effluent discharge or reuse	

Treatment			Advantages	Drawbacks	Recommendation	Full Scale applications
		Photo-Fenton (UV/Fe/H_2O_2)	-High CECs removal -Use of solar irradiation -Effective as disinfection process too	-Generation of by-products that can be toxic. -Addition of complexing agents is necessary for working at neutral pH -Need for a reactor with a specific configuration (implementation, operation and maintenance costs)	-Toxicity tests before effluent discharge	No full-scale evidence
		Ozonation (O_3)	-High CECs removal -Fast reaction with high efficiency -Low selectivity (removal of a wide-range of compounds) -Lower energy demand compared to UV/H H_2O_2. -Effective as disinfection process too	-Generation of by-products that can be toxic.	-Toxicity tests before effluent discharge -Post-treatment for by-products removal is required (i.e. biological active sand filter)	Switzerland (Swiss Water Association is planning as full-scale advanced treatment for CECs removal (www.micropoll.ch). (1)
	Heterogeneous	UV/TiO_2,	-High CECs removal -Use of solar irradiation -Effective as disinfection process too	-Generation of by-products -Need for a reactor with a specific configuration (implementation, operation and maintenance costs) -Catalyst removal is required (i.e. filtration)	-Reuse of the catalyst -Use of supported catalyst systems.	Not used at full scale. It is necessary more efficient photocatalysts at low costs.

4.5 Summary

Human and ecological demands for water resources will encourage a substantial increase in the use of recycled water for irrigation, which, however, inevitably introduces trace organics to arable soil and has the potential to contaminate food produce and exert negative effects on the ecosystem as a whole. When agricultural fields are irrigated with recycled water, CECs are unlikely to significantly accumulate in soil, as most CECs are susceptible to degradation in multiple pathways. Most pharmaceuticals and their metabolites are readily degraded in irrigated soils. On the other hand, halogenated compounds such as PBDEs, some disinfection byproducts, and perfluorinated compounds are known to be more persistent in soil environments. Studies to date have suggested that CECs introduced into soil via irrigation are mainly accumulated in the surface soil layer; only CECs with low sorption capacity and long persistence may be leached appreciably under intensive or long-term irrigation. Degradation, combined with sorption as determined by the physicochemical properties of soil and chemicals, dictate the availability of CECs for plant uptake. Once taken up, the translocation pattern of specific CECs is controlled by physicochemical properties of CECs and characteristics of plants. Studies to date have shown that plant roots are the likely plant organ with most significant accumulation of CECs, while fruits or grains with the least accumulation. Therefore, applying recycled water to specific crops such as fruit trees or non-food crops such as pasture and fodder crops could be a promising scheme to minimize human exposure.

Results from the limited field studies point to generally low likelihood for CECs to accumulate in food produce as a result of irrigation with recycled water. However, due to our incapacity to evaluate the cocktail effect of CECs as well as our poor knowledge regarding the toxicity of CEC transformation products, the actual risk may be underestimated. To maximize the use of limited resources and research capacity, a short list of CECs with the greatest potential for plant accumulation or offsite movement should be derived based on their occurrence, mobility, persistence, bioaccumulation and biological activity (Shi et al., 2022). More research is urgently needed to fill these knowledge gaps to better elucidate the fate and risks of trace-level CECs in the recycled water irrigation-soil-plant-human continuum and ultimately the exposure to humans via dietary intakes of the impacted agricultural products, as well as the ecological risk of CECs towards non-target terrestrial organisms.

CHAPTER 5

Wastewater Reuse for Irrigation: A Review of Research, Regulations, and Risks

5.1 Introduction

Increasing attention is being paid to the current global irrigation water crises with a recognition that the supply of fresh water for crop production is going to reach a critical stage in the not-too-distant future (Blumenthal et al., 2000; Rodriguez et al., 2009). Water scarcity throughout the world is well documented (Xie and Lark, 2021) with changing climate, severe droughts, and urbanization continuing to place undue environmental pressures on freshwater supplies (Bailey et al., 2018). However, shortages of fresh water sources for irrigation are not new. Alternatives have been developed to combat these pressures including desalination, reclaiming water from storm runoff, harvesting rainwater, and treating wastewater (Cao et al., 2009; Radcliffe, 2006). Recent estimates suggest that water used in agriculture accounts for greater than 65% of the global freshwater withdrawals in the United States (U.S.) and 70% worldwide (Zou and He, 2016). Because sewage is made up of over 99.8% water (Von Sperling, 2007) (Figure 1) reclaiming water from sewage is an obvious way to reduce the burden on global water supplies. The U.S. alone has over 16,000 wastewater treatment plants (WWTPs) that treat an average of 62 billion gallons of wastewater a day, whereas the European Union (E.U.) has just over 18,000 WWTPs that treat just over 40 billion gallons of wastewater a day, however; data is relatively sparse and difficult to aggregate for all countries (ASCE, 2021; Sato et al., 2013).

The use of wastewater in agriculture was documented as far back as Ancient Greece when arid regions experiencing prolonged droughts and systemic food scarcity used sewage converted to irrigation source for irrigation of crops (Angelakis et al., 2018). Yet the commonality of its use did not mean the use of untreated wastewater was without risk or did not result in illness, disease, or death. As there was relatively little knowledge about microorganisms and disease in general, much less how microorganisms proliferate in the environment, what mechanisms cause them to die off naturally, and which treatments were most effective for their removal (Angelakis et al., 2018; Blumenthal et al., 2000; Kaur et al., 2020). However, since the turn of the 20th century with the advent of WWTPs, treating sewage has advanced to the point that reuse water is considered a safe and viable source for irrigation of ready-to-eat (RTE) vegetables (Angelakis et al., 2018; Asano, 1991; Sheikh et al., 1990). And yet, even as wastewater advances continue into the 21st century, approximately 10% of the world’s population still consumes food irrigated with untreated wastewater (Hamilton et al., 2007). These regional populations employ vastly different regulatory structures to manage public health, ranging from requiring reuse water to be of drinking water quality to allowing completely unregulated use of untreated effluent (Bozkurt et al., 2021). Even among countries that conduct highly effective and efficient wastewater treatments, there remains steady debate over which treatment methods are truly successful at achieving the greatest public health outcomes, spurring multiple research studies and subsequent reviews of these studies.

To date, many thorough and extensive reviews have been published that examined the safety of different methods of wastewater treatment (Dumontet et al., 1999; Ibekwe and Murinda, 2019), the value of different regulatory thresholds (Blumenthal et al., 2000; Rock et al., 2019; Shoushtarian and Negahban-Azar, 2020), the rates of illness following exposure to partially treated and untreated wastewater, (Adegoke et al., 2018; Blumenthal and Peasey, 2002), and the way these data are used to quantify risk (Olivieri et al., 2014; Zhiteneva et al., 2020). This chapter is meant to briefly review the current knowledge of potential risks associated with microbial pathogens in treated wastewater used to irrigate fresh produce. It is not the intent of the authors to determine which methods of treatment should be applied, which pathogens should be considered of greatest concern, or which regulations should be applied. Rather, it is meant to be a discussion of the evolving guidelines governing irrigation with reuse water, potential risks from known pathogens common to produce production and recommendations for improving adoption of water reuse moving forward.

5.2 Past and Current Knowledge

5.2.1 Overview of Wastewater Reuse Guidelines Related to Food Safety in the US and Abroad

In 1918 the California State Board of Health published the first water reuse criteria which barred the irrigation of crops with raw wastewater and limited irrigation with reuse water to non-edible or cooked crops (Paranychianakis et al., 2015). Sixty years later, the California (1978) Legislature passed Title 22 and other health and safety statutes that regulated the reuse of wastewater, placing tight restrictions on the quality of water supplied by wastewater reclamation and use. In short, the statute established that effluent from WWTPs is held to strict standards (<2.2 total coliforms/100 mL), which set the groundwork for a hotly debated precedent worldwide (Angelakis et al., 2018; California Department of Health Care Services, 2001).

Following the California Title 22 requirements, a panel of experts from the World Health Organization (WHO) convened to discuss the potential health hazards associated with the use of reuse water for RTE or minimally processed produce. They acknowledged that the standards adopted in California via Title 22 were designed to eliminate risk and that the standard was 1) unachievable by many developing countries (Kamizoulis, 2008) and 2) unnecessarily restrictive given current microbiological and epidemiological data (Mara et al., 1989). Following this, the WHO Scientific Group on Health Aspects of Use of Treated Wastewater for Agriculture and Aquaculture developed microbial targets for wastewater reuse at <1000 total coliforms/100 mL (Mara et al., 1989) and a recommended target of $\leq 10^{-6}$ disability-adjusted life years (DALYs) per person per year (World Health Organization, 2006). DALYs—an academic way of saying the proportion of a year that has been lost either through death (mortality) or diminished capacity due to disease (morbidity)—are applied in recognition of the variable state of water quality throughout the world and the difficulty in adhering to particular microbiological standards (Chen et al., 2015; Kamizoulis, 2008). It was believed that the use of epidemiological data would be most valuable in determining safety in regions with higher rates of enteric illness with the simultaneous need for increased irrigation water supplies.

Yet not all standards, even within the U.S., are as strict as the California Title 22 (Rock et al., 2019), nor as achievable as the more relaxed WHO standards (WHO, 1989). Historically there has been little standardization of regulations for the reuse of water as it pertains to irrigation of RTE produce. Shoushtarian and Negahban-Azar (2020) reviewed 70 regulations and guidelines from around the world and found that while human health was the primary focus of these regulations, the disparity between the standards and levels of public health protection was significant (Dingemans et al., 2020). In recognition of these disparities, the E.U. laid out guidance on the use of reclaimed water for agricultural purposes stopping short of requiring quantitative microbial standards, instead of relying on risk assessments and process controls within wastewater treatment facilities (Council of the European Union, 2020). Risk management provisions are included to assess and address potential health and environmental risks, as well as permitting requirements. Meaningful implementation of the E.U. framework for the regulation of wastewater reuse has been hampered by criticism; it is considered both overly ambitious while also failing to acknowledge certain challenges like antibiotic resistance and emerging diseases (Dingemans et al., 2020).

Unlike the E.U., and regardless of the seminal standards established in California, water reuse in the U.S. is still not covered under a federal standard or shared directive (Rock et al., 2019). However, the United States Environmental Protection Agency (USEPA) developed, "Guidelines for Water Reuse" under the Clean Water Act, as a road map for individual states to implement water reuse programs, stopping short of regulating microbial thresholds as a national policy (USEPA, 2012a). In 2011, the Food Safety Modernization Act (FSMA) was passed by Congress in response to a growing number of American foodborne outbreaks. Four years later the U.S. Food and Drug Administration (USFDA) began to implement portions of FSMA, that require agricultural water to be "of safe and of adequate sanitary quality for its intended use (Rock et al., 2019; USFDA, 2011). FSMA also requires produce growers to test their water for microbial contamination if the grower is near or adjacent to risky land-use practices, such as WWTPs or confined animal feeding operations. The USFDA borrowed the USEPA 2012 Recreational Water Quality Criteria, (E. coli geometric mean <126 CFU/100 mL, statistical threshold value <410 CFU/100 mL) as a standard for growers to achieve based on how water is delivered (USFDA, 2011). However, this standard has been broadly challenged as both being burdensome to produce growers and ineffective at preventing illness in consumers of RTE (Gradl and Worosz, 2017; Rock et al., 2019).

While the shared goal of these various standards is either the reduction of foodborne illness or the maintenance of illness rates at acceptable levels, they do not in and of themselves require the absence of or reduction in pathogens. This means that even though they know which pathogens are responsible for the highest rates of foodborne illness they do not have data on the prevalence or concentrations of these pathogens in wastewater streams.

In some cases, regulations could be a challenge and burden for water reuse, as for example in the case of very restrictive requirements based on the precautionary principle. For example, the health risk-based regulations for irrigation, such as those developed in Australia and used as the basis for the new European regulations, require an additional health risk assessment

(Qualitative or Quantitative Microbial Health Risk Assessment, QMRA) and validation of log removal of treatment technologies, in addition to water quality monitoring.

5.2.2 Understanding Pathogen Risks

Enteric pathogens are agents of disease that are excreted in the feces of animals and humans. They can be bacteria, viruses, protozoal parasites, helminths, or even fungi. They are frequently associated with exposure to contaminated water, consuming contaminated food, or coming into contact with the feces of infected individuals. Enteric pathogens are easily spread via waterborne routes and so the potential for contamination of irrigation water supplies is of primary concern in agricultural regions, particularly where RTE commodities are grown (Atwill et al., 2012).

However, the prediction of the true risk of enteric disease following irrigation with reuse water must consider several factors: 1) reuse water must contain pathogens that are infective to humans; 2) the pathogens must be in high enough concentrations to likely lead to disease, and 3) a susceptible person must come in contact with the pathogen at the disease-causing concentration (dose). The concentration and type of pathogen (bacterial, viral, or parasitic) depends on the morbidity (rate of disease) of various enteric illnesses by the people contributing to the wastewater system (Cooper, 1991). Meaning users of a wastewater system must be shedding those pathogens in their feces. These rates are not evenly distributed across the world, nor are all people equally susceptible (Kirk et al., 2015). The sanitation practices, dietary habits, and environmental exposure of the community greatly influence the likelihood of pathogens entering the waste stream (Cooper, 1991).

5.2.3 Quantifying Microbiological Risks

How regulatory bodies define or quantify potential risks to consumers is highly variable and dependent upon not only the population of consumers under question but the underlying prevalence of disease in the population and the resources necessary to reduce that risk (Lammerding and Paoli, 1997). Risk is not static and frequently bears seasonal characteristics (Bottichio et al., 2020; Bozkurt et al., 2021; Sharma et al., 2020; USFDA, 2019). Further, though many of the guidelines for irrigation water safety are said to be 'risk-based', it remains unclear how monitoring parameters were chosen and how they fit into management decisions about applications of certain thresholds for different uses of reuse water while protecting public health (Troldborg et al., 2017).

Health risks following exposure to pathogens are typically quantified in one of two ways epidemiological studies or quantitative microbial risk assessment models (QMRA).

Epidemiological studies use the incidence of disease in a population and their known (or assumed) exposures to the pathogen or food item under question to calculate the probability of disease for future exposures. The purpose of these types of studies is to find statistical associations between well-defined exposure routes/levels and incidence of disease with the hope that may then be applied more broadly. However, determining the risk of disease after exposure requires a population of people to first be exposed—either voluntarily or incidentally— thereby incurring a yet undefined risk of becoming ill, perhaps severely. For this reason, epidemiological estimations of risk in one region must frequently borrow from studies conducted elsewhere resulting in inaccurate risk estimates (Mara et al., 2007). Applying this approach to agriculture has added disadvantages, with the potential differences in growing practices, treatment programs, and general population health, among others (Hamilton et al., 2007).

In contrast, the QMRA approach is used to model risk profiles for specific scenarios following four distinct steps 1) hazard identification, 2) exposure assessment, 3) dose-response modeling, and 4) risk characterization (Rose and Gerba, 1991). Human health risks posed by microorganisms in wastewater have been estimated by QMRAs since the 1980s (Haas et al., 2014). The outcomes of these studies are frequently expressed in terms of cases of illness per defined population size (typically 1,000-100,000) or through the estimation of DALYs (Chhipi-Shrestha et al., 2017a; World Health Organization, 2006). However, QMRAs require large amounts of data to give realistic, if not always accurately predictable, probabilities of risk associated with produce consumption following wastewater irrigation. Paradoxically, these data generally come from previous epidemiological studies. Both approaches have validity, and both require the definition of the pathogen of concern and potential exposures; topics covered below.

5.2.4 Bacteria, Viruses and Protozoa of Primary Concern

A foodborne disease outbreak is defined as the occurrence of two or more cases of an illness associated with the consumption of a common item (Brown et al., 2017). In a discussion on potential implications and risks associated with the use of reclaimed water it is important to understand which potential pathogens are of primary concern based on virulence (likelihood of causing disease) but also their association with irrigation water supplies, and subsequent produce consumption. While there are a large number of potentially pathogenic organisms to be found in wastewater streams (USEPA, 2012a), a relatively small number of them have been associated with produce-related outbreaks of foodborne disease (Sivapalasingam et al., 2004) and fewer still have been tied to contaminated irrigation supplies (Bottichio et al., 2020; Gelting et al., 2011). As any discussion on risk must incorporate the identification of the causative agent of a disease, below they briefly examine the most common, and most detrimental, pathogens associated with RTE produce consumption in the U.S. and worldwide.

5.2.4.1 Bacteria

Widespread outbreaks of foodborne disease and gastrointestinal illness following consumption of contaminated produce have been regularly attributed to enteric bacterial pathogens, most commonly *Escherichia coli, Salmonella spp.,* and *Listeria monocytogenes* (Griffin and Tauxe, 1991; Heiman et al., 2015; Krishnasamy et al., 2020). Contaminated irrigation supplies have frequently been implicated, or at least suspected, as the source/vehicle of contamination of nearby produce fields (Gelting et al., 2011). However, bacterial pathogens are ubiquitous in surface water supplies throughout the U.S. (Atwill et al., 2012; Partyka et al., 2018; Partyka et al., 2016; Sharma et al., 2020; Weller et al., 2015) and the world (Kirk et al., 2015; Lopez-Galvez et al., 2016).

E. coli O157:H7 (O157) was first identified in the early 1980s as being the causative agent of a distinctive syndrome that has the potential to cause life-threatening hemorrhagic diarrhea or, in extreme cases, hemolytic uremic syndrome (Griffin and Tauxe, 1991). While several other serotypes of E. coli have been identified that share a similar pathogenic potential (e.g., O26, O121, O145)—collectively referred to as enterohemorrhagic (EHEC) or Shiga-toxin producing E. coli (STEC)—O157 remains the most common and the most studied of the group (Griffin and Tauxe, 1991; Heiman et al., 2015). From 1982-2018 there were nearly 120 outbreaks of O157 associated with the consumption of fresh produce in the U.S. alone (Interagency Food Safety Analytics Collaboration, 2020; Rangel et al., 2005).

Research on the O157 infections following leafy green consumption began with increased urgency after 2002 when a series of multi-state outbreaks in the U.S. received heightened media coverage; a single outbreak in spinach during 2006 resulted in 205 illnesses and 3 deaths (Cooley et al., 2007), not to mention a multi-month recall of bagged spinach and financial damage to the industry. Investigations into the outbreaks were keenly focused on potentially contaminated irrigation supplies (Gelting et al., 2015; Gelting et al., 2011). None of the farms implicated in these outbreaks were irrigating with reclaimed wastewater; however, the potential for contaminated irrigation water as a plausible cause was investigated and the possible connection contributed to renewed calls for rigorous on-farm irrigation testing (LGMA, 2020). Regardless of increased attention to the risks and risk-based testing programs, non-reuse irrigation water was implicated in two recent outbreaks of O157 in Romaine lettuce leading to the death of 5 people (Bottichio et al., 2020; Hoff et al., 2019).

Salmonella spp. is also a well-known and well-studied bacterial organism with multiple serotypes that are responsible for more than 1.2 million cases of gastroenteritis and 450 deaths in the U.S. annually (Bosch et al., 2016). Salmonella spp. was responsible for over 78 million cases and more than 28 thousand deaths worldwide in 2010 alone (Kirk et al., 2015) and remains one of the most important causal agents of foodborne illness in developed countries (Santiago et al., 2018). Salmonella spp. was the most commonly identified etiological agent of produce-related outbreaks in the U.S. from 1973-1997 (30/103, 29%) (Sivapalasingam et al., 2004) and remains an important contributing factor to this day. Over 30% of outbreaks of salmonellosis (276/905) were attributed to varying types of produce (seeded vegetables, fruits, or leafy greens) from 1998 to 2018 (Krishnasamy et al., 2020).

Listeria monocytogenes rarely causes foodborne illness in the U.S. but is a leading cause of death from foodborne illness in the U.S. with an estimated case fatality rate of 17% (McCollum et al., 2013). Listeriosis is particularly risky for the elderly, the immunocompromised, children, and pregnant women; infections have been linked to fetal abortion (McCollum et al., 2013). L. monocytogenes was responsible for one of the deadliest foodborne outbreaks in U.S. history when, in 2011, 147 individuals fell ill and 33 died from eating contaminated cantaloupe (McCollum et al., 2013). From 1998 through 2018, *L. monocytogenes* was identified as the etiological agent in 44 produce-associated outbreaks in the U.S. (Interagency Food Safety Analytics Collaboration, 2020). Globally, it is estimated that listeriosis was responsible for >23,000 cases of illness and over five thousand deaths during 2010 (de Noordhout et al., 2014)

Listeria spp. is ubiquitous in the environment and is regularly isolated in produce production regions, including on-farm water supplies (Strawn et al., 2013; Weller et al., 2015; Zhu et al., 2017). However, though *Listeria spp*., and other bacterial pathogens, may persist in irrigation water supplies (Gu et al., 2021; Sharma et al., 2020; Strawn et al., 2013; Weller et al., 2015), contamination of irrigation water has yet to be clearly associated with outbreaks of listeriosis.

5.2.4.2 Viruses

Viral pathogens survive relatively long periods in water, are resistant to some treatment processes (Lopez-Galvez et al., 2016; Rose and Gerba, 1991), and cause a large number of illnesses annually (Pearce-Walker et al., 2020). A recent survey found that viruses accounted for 68% of foodborne outbreaks in the U.S. between 2009 and 2013 (Brown et al., 2017). Among the enteric viruses present in reuse water, norovirus (NoV) is the leading cause of foodborne gastroenteritis in people of all ages worldwide with an estimated number of 125 million cases in 2010 (Kirk et al., 2015).

Irrigation water has the potential to be an important vector for the transmission of common viruses (Gonzales-Gustavson et al., 2019; Lopez-Galvez et al., 2016; Rusinol et al., 2020a). Multiple studies have found higher occurrences of enteric viruses in reclaimed water supplies compared to other sources of irrigation (Rusinol et al., 2020a; Truchado et al., 2021), suggesting they possess a higher potential risk of crop contamination than more frequently studied produce-borne pathogens (e.g., E. coli O157 and L. monocytogenes) (Rusinol et al., 2020a; Rusinol et al., 2020b). However, the lack of monitoring for the presence of viruses in irrigation water prevents the comprehensive understanding of their impact on public health via produce consumption. Recently, many research groups have been able to detect macromolecules (ribonucleic acid or RNA) of the SARS-CoV-2 virus in wastewater which can be used to monitor COVID-19 in a community (Kitajima et al., 2020). Also, has been suggested that conventional wastewater treatment is adequate to control the transmission of COVID-19 as RNA fragments of SARS-CoV-2 have not been detected in fully treated sewage (WHO 2020).

5.2.4.3 Protozoa

Giardia and Cryptosporidium are some of the most common parasites in surface water supplies in agricultural environments (Drummond et al., 2018), passing zoonotically from animals to humans and vice versa (Atwill et al., 2012). Unlike viruses and bacteria, these waterborne parasites can lay dormant in cysts (Giardia) or oocysts (Cryptosporidium) for weeks to months in the environment (Atwill et al., 2012). *Cryptosporidium parvum* is the second most common waterborne pathogen worldwide, next to NoV (Kirk et al., 2015), with an estimated 30,000 cases of cryptosporidiosis occurring annually in the U.S. (Yoder et al., 2012). C. parvum is highly infectious in healthy adults, (Atwill et al., 2012; Drummond et al., 2018); however, the elderly, young children, and immunocompromised individuals are particularly susceptible to cryptosporidiosis (Rossle and Latif, 2013). *Giardia* duodenalis is the enteric parasite responsible for the most incidents of parasitic diarrheal disease in the U.S. (Conners et al., 2021) Globally, Giardia led to fewer produce-associated illnesses and deaths than Cryptosporidium (Kirk et al., 2015); however, intestinal parasites can spread easily from person to person, and the infected individual may not show symptoms, so cases may not be attributed to food or produce consumption.

5.2.5 Defining Exposure

Exposure, in terms of pathogens, is often difficult to estimate with any level of accuracy. First, one must determine the probable pathogen loads in irrigation supplies, then determine the concentration of pathogens transferred to the edible portion of a commodity, followed by estimations of pathogen die-off post-irrigation, and ending by estimated rates/volumes of consumption in a given population. All these estimates are likely to vary by region and population. Current estimates of pathogens both entering and exiting the wastewater stream can vary based on not only the actual prevalence of pathogens, but also treatment methods, volumes sampled, and detection capabilities (Harwood et al., 2005) (Table 5-1).

Table 5-1. Efficacy of WWTPs for Removal of Microbial Pathogens after Tertiary Treatment.
Source: Adapted from *Science of the Total Environment* 828(2022); by M. Partyka and R. Bond; "Wastewater Reuse for Irrigation of Produce: A Review of Research, Regulations, and Risks"; p. 154385; Copyright (2022), with permission from Elsevier.

Pathogen Group	Influent Concentration	Effluent Concentration	Average Reduction
Bacteria			
E. coli	6.4 $\log_{10}$/100 mL† 10^{5}–10^{10}††	0.05 $\log_{10}$/100 mL†	2.0-6.0 log††
Salmonella spp.	4.1 $\log_{10}$/100 mL† 10^{3}–10^{5}††	−0.86 $\log_{10}$/100 mL†	2.0-6.0 log††
Viruses			
Norovirus	3.49 $\log_{10}$/100 mL†	0.24 $\log_{10}$/100 mL†	~1.5-2.13 log†††
Adenovirus	4.57 $\log_{10}$/100 mL† 10-10^{4}††	2.72 $\log_{10}$/100 mL†	>1.0 log††
Protozoa			1.0-3.0 log††
Cryptosporidium	1.57 $\log_{10}$/100 mL† 0-10^{4}††	−0.78 $\log_{10}$/100 mL†	0-3.0 log††
Giardia	1.48 $\log_{10}$/100 mL†	−1.22 $\log_{10}$/100 mL†	0.5-3.0 log††

† (Bailey et al., 2018) †† (Health Canada, 2010) ††† (Flannery et al., 2012)

In fresh produce, microbial pathogen contamination is usually present in very low levels (<1%) (Van Pelt et al., 2018) and unevenly distributed, further decreasing the probability of detection (Allende et al., 2018). While studies have been performed to estimate the transfer of pathogens to multiple produce types (e.g., leafy greens, zucchini, and nectarines) (Lopez-Galvez et al., 2016; Mok and Hamilton, 2014; Vivaldi et al., 2013) and through different irrigation methods (e.g., drip, furrow, and overhead sprinkler) (Allende and Monaghan, 2015; Allende et al., 2018), there is no universal understanding of the frequency of pathogen transfer, and adherence to RTE produce.

Part of the difficulty in achieving understanding is due to the many estimates and assumptions that are needed to capture the probability of waterborne pathogen transfer. For instance, one must first measure, or estimate, the amount of water captured and/or retained by a specific commodity following irrigation; a value that can be highly variable. For example, Mok and

Hamilton (2014) found that overhead irrigation of different leafy green vegetables—including lettuces—captured volumes of water ranging from a median of 0.01 mL/g to 0.06ml/g, while other QMRA studies have estimated capture by lettuce to be as high as 0.11 mL/g (Shuval et al., 1997). Even if estimates of water retention were in complete agreement, they may not correlate with pathogen retention on the produce surface.

Not surprisingly, the probability of transferring pathogens from contaminated water to a commodity decreases when water is prevented from coming into direct contact with a commodity. Multiple studies have found that of the three main irrigation types, overhead sprinkler irrigation has the highest associated risk, followed by furrow irrigation and finally the surface/subsurface drip irrigation which delivers irrigation water (Bernstein and Sacks, 2011; Rock et al., 2019; Troldborg et al., 2017; Truchado et al., 2016). However, it is worth noting that furrow irrigation may also impose additional, non-consumptive, risks to field workers that may come in direct contact with water supplies (Adegoke et al., 2018; Blumenthal and Peasey, 2002).

Although water retention estimates are necessary to model risks following vegetable consumption in QMRA studies, variation in consumption rates is likely to have a greater influence over the modeled risk probabilities than water retention (Mok and Hamilton, 2014). For example, the consumption of lettuce can vary dramatically between cultures, with an estimated consumption of 21.81 g/per person/day in Australia (Bozkurt et al., 2021) and up to 171.94 g/per person/day in China (Mok and Hamilton, 2014). This discrepancy could mean the difference between models predicting water as being safe to use and the alternative. Therefore, for QMRA approaches to have the most utility they should try to incorporate local consumption data. In recognition that exposure remains one of the trickiest components for QMRAs, one study employed the use of Quantitative Microbial Exposure Assessments (QMEAs) as an augmentation to current QMRAs to improve exposure modeling throughout the farm-to-fork continuum (Allende et al., 2018).

In summary, exposures defined in QMRA studies require a large amount of, preferably localized, data on the prevalence of pathogens, expected concentrations when present, irrigation application methods/rates, factors contributing to pathogen reduction, multiplication, or growth potential, and eventual consumption values. (Allende and Monaghan, 2015; Allende et al., 2018). However, in absence of local data, it may be necessary to develop more conservative estimates of risks and or work to improve methods of risk mitigation (Mok and Hamilton, 2014).

5.2.6 Pathogen Reduction through Wastewater Treatment

Wastewater treatment facilities employ multiple stages of treatment to remove contaminants and purify effluent. Generically, the primary treatment stage screens for large debris and sediment; secondary treatment uses biological processes to further break down organic and inorganic material; and finally, tertiary or advanced treatment uses physical and chemical processes to break down and sequester potential contaminants prior to release as effluent (Sonune and Ghate, 2004). Removal of pathogens in these processes occurs principally at the secondary treatment stage, where microorganisms are utilized to remove contaminants aerobically and anaerobically.

Most secondary treatment is achieved through activated sludge, a revolutionary advancement developed in 1904 (Asano, 1991). The sludge relies on biological floc (biofilm) formation which then settles out removing solids and clarifying effluent (Asano, 1991; Scholz, 2016). Tertiary treatment utilizing various methods, alone or in combination (e.g., chlorine, ozonation, activated carbon, or ultraviolet radiation) further degrades microorganisms to varying degrees (Sonune and Ghate, 2004) .

In the late 1970s, the Pomona Virus Study (PVS) was conducted in Los Angeles, California, to assess the effectiveness of a suite of tertiary treatments for the minimization of pathogens in effluent supplies. Data revealed that the use of either filtration or carbon absorption in the tertiary stage returned effluent to viral loads most similar to potable water treatment (Dryden et al., 1979). The PVS was followed by the Monterey Wastewater Reclamation Study for Agriculture (MWRSA) (Sheikh et al., 1990). The decade-long MWRSA focused specifically on addressing potential risks from using treated wastewater to irrigate produce intended to be consumed raw, a use that had been previously either disallowed or discouraged by California public health statutes.

These studies provided the technical basis by which regulations would eventually be developed to promote the safe and effective use of reclaimed water for the irrigation of RTE produce (Asano, 1991). However, in the succeeding decades, questions continue to be asked about the efficacy of the large variety of treatment approaches for reducing all types of pathogens, particularly in areas of the world that are not governed by the strict regulatory standards or pathogens proven to be resistant to standard treatments.

Regardless of the success of the early studies in California, the production of reclaimed water using activated sludge processes, filtration, and disinfection is not universally effective for removing pathogens. Several factors, like disinfectant concentration, exposure time, and pathogen resistance to treatment, are involved in the success of the tertiary treatment (Hijnen et al., 2006; Santiago et al., 2018). While wastewater treatment systems can commonly eliminate 20–80% of enteric viruses (Ottoson et al. 2006; Barker-Reid et al. 2010; Mok and Hamilton, 2014), (oo)cysts of protozoal pathogens are typically resistant to many conventional disinfection steps, particularly chlorination, resulting in unsatisfactory reductions (Bailey et al., 2018). In 2015, Allende and Monaghan summarized the most commonly applied water treatments for agricultural water. Physical and chemical disinfection systems have been explored as methods to remove human pathogens from agricultural water sources, although disinfection treatment of irrigation water is still a very limited practice.

5.2.6.1 Indicators of Success

The microbiological safety of irrigation water supplies, regardless of the source, is most frequently assessed through the presence of bacterial indicators (total coliforms, fecal coliforms, and/or *E. coli*). In the U.S. the proposed indicator threshold used for agricultural water criteria under the Produce Safety Rule uses a long-established body-contact recreation standard that was developed based on epidemiological studies of beachgoers at freshwater beaches (Rock et al., 2019). Similarly, the somewhat less stringent WHO guidelines (a combination of microbiological criteria and maintenance of disease burden) and California's most-rigorous Title 22 criteria are also based on epidemiological studies and QMRAs (Blumenthal et al., 2000; Mara et al., 1989; Olivieri et al., 2014).

While these bacterial indicators have well-established histories as part of the regulatory underbelly of water quality monitoring (USEPA, 2012b) their efficacy has been hotly debated (Cooper, 1991; Wen et al., 2020), particularly for monitoring treated wastewater. For example, Chern et al. (2013) found that E. coli concentrations in treated wastewater samples were strong indicators of treatment efficacy while Rock et al. (2015) acknowledged that while microbial indicators may not correlate with specific pathogens, their concentration has been linked epidemiologically to illness probabilities. Further, human health risk estimates from a single case study that investigated three wastewater treatment facility effluents in Canada recommended pathogenic E. coli as the bacterial indicator most sensitive for use in all water reuse categories (Chhipi-Shrestha et al., 2017b).

Those examples aside, many more studies have roundly criticized the use of bacteriological standards to protect against either viral or protozoal infections. Studies on the presence of viral and protozoal pathogens in treated wastewater demonstrate that low concentrations of bacterial indicators do not necessarily indicate the absence of non-bacterial pathogens (Lopez-Galvez et al., 2016). For example, Rose (2005) and Bonetta et al. (2016) report that samples with no detectable *E. coli* were found to have significant concentrations of potentially pathogenic viruses or protozoal parasites.

Additionally, multiple studies have found that even bacterial pathogens are sometimes not controlled by treatments that reduce bacterial indicators, particularly species known to regrow/proliferate after wastewater treatment (Caicedo et al., 2016; Ibekwe and Murinda, 2019; Jjemba et al., 2010). For example, Kulkarni et al. (2018) compared the bacterial communities of influent and effluent samples from four WWTPs in two regions of the U.S. and found that while bacteria associated with the human gut microbiome were significantly reduced in all effluent samples, potential pathogens from the genera Legionella, Clostridium, and Mycobacterium were found in greater abundance in post-treatment effluent samples.

Even as the use of microbial indicators continues to be debated, it is worth remembering that microbial criteria are not strictly based on the association of indicator organisms with a particular pathogen but rather the probability of illness once the indicator threshold is exceeded (USEPA, 2012b).

5.2.6.2 Are Global Standards Useful?

Different microbiological guidelines for water reuse have different pathways for protecting public health (Shoushtarian and Negahban-Azar, 2020); whether it's maintaining indicator concentrations below defined thresholds, the prevention of excessive cases of enteric disease, or ensuring the modeled risk remains below an acceptable limit (Blumenthal et al., 2000). This seems to suggest that regulatory authorities need only choose a route towards the common goal of public health based on policy preference or technical ability. Yet at the regional level, the absence of unified or at least relatively comparable water reuse regulations and guidelines may result in uncertainty among stakeholders (e.g., farmers, consumers, and policy makers), thereby slowing down the promotion of water reuse in agriculture (Shoushtarian and Negahban-Azar, 2020).

And yet, while greater standardization at the regional/local level may be useful, available data do not suggest the need for globalized standards for pathogen or indicator reduction in wastewater effluent. Rather, standards should be risk-based to be effective. The risk from wastewater reuse for produce irrigation depends on a multitude of factors, such as irrigation method and any post-harvest processes before consumption (Mok and Hamilton, 2014). If the risk is variable then monitoring strategies and standards should also be variable and be tailored toward the etiological agents of disease that dominate regional water supplies, common growing practices, and produce consumption rates that define exposure risk. The calls for the creation of universal standards for wastewater monitoring ignore decades of epidemiological research regarding exposure risks and potentially jeopardize the broadscale use of reuse water in the future (Havelaar, 2001; Hespanhol and Prost, 1994).

5.3 Future Research Directions and Recommendations

5.3.1 Stakeholder Engagement Necessary for Broad Adoption

To minimize the human health risks from unsafe wastewater irrigation, the WHO's related 2006 guidelines suggested a broader concept than the previous (1989) edition by emphasizing, especially for low-income countries, the importance of risk-reducing practices from 'farm to fork'. Another challenge concerns local capacities for quantitative risk assessment and the determination of a risk reduction target. Being aware of these challenges, the WHO has invested in a sanitation safety planning manual which has helped to operationalize the rather academic 2006 guidelines, but without addressing key questions, e.g., on how to trigger, support, and sustain the expected behavior change, as training alone is unlikely to increase the adoption of health-related practice (Drechsel et al., 2022).

The main challenge of wastewater irrigation is the common reality of its unplanned use in urban and peri-urban areas. Beset by water challenges, especially in arid regions where the land is plentiful, but water resources are not, agricultural use of reclaimed water is an ecologically viable solution to combat worsening droughts and overtaxed aquifers. (Bailey et al., 2018). To date, no conclusive evidence has been found to implicate tertiary treated reuse water as a risk factor for produce-related illness or outbreaks (Orlofsky et al., 2016). Further, data seem to suggest that—provided that regulatory standards and practices are being adhered to—using treated reuse water to irrigate produce poses no greater risk to consumers than other sources of irrigation water (Blumenthal et al., 2000; Bozkurt et al., 2021; Busgang et al., 2015; Lopez-Galvez et al., 2016; Obayomi et al., 2019; Mohr et al., 2020; Mori et al., 2020; Ofori et al., 2021). In fact, the State of California's adherence to Title 22 is largely credited with providing public health protection to wastewater reuse as there have been no known foodborne outbreaks or increased incidences of illness attributed to wastewater reuse in agricultural settings since its inception (Olivieri et al., 2014).

Still, perceptional gulfs exist between the public and water resources managers on the use of reuse water for produce production; the former continues to struggle with the "yuck factor" stemming from a perception of elevated risk while the latter—recognizing the inherent safety and relatively low risks—are more concerned about increasing water availability for food production (Leong and Lebel, 2020). Produce growers tend to lie in the middle of these views—they understand the inherent need to increase water resources but are concerned about the challenges of marketing produce grown from reuse water (Peters and Goberdhan, 2016; Suri et al., 2019). Globally, reuse water has garnered wide acceptance. Acceptance can be further broadened by increasing the transparency about the strengths and limitations of treatment technologies through public outreach programs (Peters and Goberdhan, 2016). Mara et al. (1989) recommended the following to improve public support while maintaining health standards:

1. Rules for reuse water application, water quality criteria, and monitoring must be in place to ensure the health and safety of not only produce consumers but also farmworkers.
2. There must be public stakeholder engagement to promote buy-in for wastewater reuse projects, including full transparency and access to real-time information.
3. Risk assessments should be completed under local, or regional, conditions to improve exposure estimates and illness probabilities.

Identification and implementation of preventive measures should be based on the **multiple barrier principle**. According to this principle, multiple preventive measures or barriers are used to control the risks posed by different hazards, thus making the process more reliable. The strength of this principle is that a failure of one barrier may be compensated by an effective operation of remaining barriers, thus minimizing the likelihood of hazards passing through the entire system and being present in sufficient amount to cause harm to public or environmental health (Alcalde-Sanz and Gawlik, 2017). However, agricultural environments are enormously complex and with the addition of emerging diseases, new technologies, and rapidly increasing environmental pressures, scientists must simplify the discussion around risk so that policymakers, produce growers, and consumers can make informed decisions (Mara et al., 2007).

However, agricultural environments are enormously complex and with the addition of emerging diseases, new technologies, and rapidly increasing environmental pressures, scientists must simplify the discussion around risk so that policymakers, produce growers, and consumers can make informed decisions (Mara et al., 2007).

5.3.2 Forming a Picture of Risk

Even with all the debates, caveats, and calls for additional research, in the more than 100 years since the California Board of Public Health allowed the use of wastewater for non-irrigation purposes and the more than 40 years since Title 22 first legislated the safe use of wastewater reuse for RTE commodities, they have learned a great deal about risks of wastewater reuse. We've found that the identification of risks to consumers depends heavily on multiple exposure criteria, much of which is highly variable from region to region and culture to culture (Allende et al., 2018; Bozkurt et al., 2021; Mok and Hamilton, 2014). They have also learned that while the most restrictive standards they create may reduce the perceptions of risk they do not prevent produce-related outbreaks from being attributed to irrigation supplies (Bottichio et al., 2020;

Hoff et al., 2019). Multiple worldwide epidemiological studies have revealed that untreated, and even partially treated, wastewater can pose significant risks to not only produce consumers but to agricultural workers, their families, and even residents and passersby (during active irrigation) (Adegoke et al., 2018; Blumenthal and Peasey, 2002). Setting unobtainable requirements for water reuse does not decrease the global health burden of enteric disease, address water shortages, or help to increase crop production in the parts of the world that are also struggling with poverty and food scarcity (Blumenthal and Peasey, 2002; Kirk et al., 2015). In short, the assumptions of risk are personal (water management, farmer, and consumer) and should be considered carefully before water reuse is either allowed or disallowed in produce production environments.

5.4 Summary

The burden of disease caused by the contamination of ready-to-eat produce with common waterborne microbial pathogens suggests that irrigation supplies should be closely monitored and regulated. Simultaneously freshwater resources have become increasingly scarce worldwide while global demand continues to grow. Since the turn of the 20th century with the advent of modern wastewater treatment plants, reuse of treated wastewater is considered a safe and viable water source for irrigation of ready-to-eat vegetables. However strict, and often costly, treatment regimens mean that only a fraction of the world's wastewater supplies are being put to reuse. The purpose of this review is to explore the available literature on the risks associated with reuse water for ready-to-eat produce production including different approaches to reducing those risks as the demand for reuse water increases. It is not the intent of the authors to determine which methods of treatment should be applied, which pathogens should be considered of greatest concern, or which regulations should be applied. Rather, it is meant to be a discussion of the evolving guidelines governing irrigation with reuse water, potential risks from known pathogens common to produce production, and recommendations for improving adoption of water reuse moving forward. To date, there is little evidence to suggest that adequately treated reuse water poses more risk for produce-related illness or outbreaks than other sources of irrigation water. However, multiple epidemiological and quantitative risk assessment models suggest that guidelines for the use of reuse water should be regionally specific and based on local growing practices, available technologies for wastewater treatment, and overall population health. Though research suggests water reuse if generally safe, the assumptions of risk are both personal and of public interest, they should be considered carefully before water reuse is either allowed or disallowed in produce production environments (Partyka and Bond, 2022).

CHAPTER 6

Case Studies

6.1 Australia

In Australia, irrigated agriculture used about 6,300 hm^3 of water in 2019-20, of which 124 hm^3 was recycled water obtained from off-farm sources The first significant use of wastewater for agricultural production occurred in Adelaide, South Australia which in 1879 had built an underground water-borne sewerage system that discharged domestic effluent to land at a 190 ha sewerage farm at Islington, north of the city, producing forage by a cut and carry system for cattle feeding. Melbourne, a city then of about 500,000, developed its Werribee Sewage Farm from 1892, operating on a land and later a grass filtration system for grazing by beef cattle sold only for slaughter, to reach a farm size of 10,850 ha before evolving into a modern tertiary Wastewater Treatment Plant (WWTP) producing recycled water for industrial and agricultural use. As sewage systems developed across the country, towns would use the effluent from their treatment facilities, usually oxidative ponds, to irrigate public amenities at times when the public were excluded. Much of this recycled water was also used in the WWTP facility itself. By 2003, over five hundred WWTPs across Australia were engaged in the recycling of at least part of their treated effluent.

At about the same time as publication of the *Ecologically Sustainable Development Report* in 1991, the Australia states began establishing environment protection agencies. The potential damage caused by inadequately treated sewage effluent being discharged to oceans, rivers and estuaries was recognized. Regulations were brought in setting standards for discharges. Sewage Treatment Plant operators were increasingly required to come up with environmental management strategies for their discharges. Water recycling was given attention by the newly established Environment Protection Authorities (EPAs) which imposed stricter water compositional standards on the discharge of treated sewage effluents from WWTPs to receiving waters. This resulted in increased interest by WWTP operators in recycling effluent for productive purposes on land as an alternative to installing expensive biological nutrient removal plants.

In 1992, in recognition of the need to better manage natural resources, Ministerial Councils comprising Commonwealth and States/Territories Ministers for agriculture, the environment and conservation endorsed the development and implementation of a National Water Quality Management Strategy (NWQMS). Although the adopted details differed between states, there were essentially four categories of recycled water. These typically involved Class A (Highest Quality) meeting 10 *E. Coli* org/100 ml; Turbidity < 2NTU (24 hr median), < 5 NTU (max)<10 mg/L BOD; <5 mg/L Suspended Solids, pH 6 – 9 (90 percentile), Cl_2 residual: >1 mg/L after at least 30 minutes contact time where human contact, <1mg/L at point of use.<10 *E coli* /100 ml, < 1 helminth /L, ;<1 protozoa / 50 L, & <1 virus / 50 L suitable for food crops eaten raw and water reticulated to households for non-potable use., Lowest quality was class D<1 000 *E. coli* org/100 ml.<20 mg/L BOD, <30 mg/L Suspended Solids, pH 6 – 9 (90 percentile), suitable for Agriculture: Non-food crops including instant turf, woodlots and flowers.

A strategic framework for the reform of the Australian water industry was agreed in 1994 at the Council of Australian Governments (COAG) comprising the Prime Minister, state Premiers, territory Chief Ministers and a representative of the Australian Local Government Association. The reforms included separating ("unbundling") the ownership of land from rights to water, with both becoming separately tradable. During 2004–2006, the Commonwealth and the States and Territories progressively signed a 108 clause *Intergovernmental Agreement on the National Water Initiative* (NWI). The agreement included water entitlements, water markets and trading, water pricing, and management of environmental water. A component was to develop a set of guidelines to cover the potential use of recycled water, storm water and Managed Aquifer Recharge (MAR). This was achieved under the auspices of Commonwealth, States/Territories Ministers for Agriculture, the Environment and Health, through the development of Guidelines for Water Recycling, including Augmentation of Drinking Water Supplies. These Guidelines were adopted into States/Territories legislation and regulations as necessary.

The major potential source of recycled water arises in the capital cities, but they have developed in different ways. Adelaide and Melbourne have areas of land close to the cities suitable for the growth of high value horticultural crops, Brisbane developed the Western Corridor Scheme in which three major WWTPs can produce recycled water suitable for drinking, to be piped as potable recycled water to the Wivenhoe dam, Brisbane's primary water source. Growers in the nearby Lockyer Valley have sought access to this water, but it has been uneconomic, Perth is strongly dependent on groundwater and has developed a groundwater replenishment scheme for its recycled water, Sydney has limited agricultural land close to the city, but some agricultural recycled water use south of the city. There is limited recycled water available in Sydney, and much is used to offset diversions of environmental flows, In any case, Sydney discharges ninety percent of its WWTP effluent after only primary treatment into deep ocean outfalls.

The first planned major Australian reuse scheme for agricultural production was the Northern Adelaide Plains Scheme, where water from the Bolivar WWTP, which had replaced its century-old Islington land treatment facility, was made available for high value vegetable production. As originally built, Bolivar's secondary treated effluent had been discharged to Gulf St Vincent. During the period 1949 to 1995, approximately 40 km^2 of seagrass was lost along the Adelaide coastline. High nutrient sewage was considered a significant cause of this loss, and the Adelaide metropolitan WWTPs were required to develop Environmental Improvement Plans to upgrade the quality of their effluents if they were to continue discharging to ocean. To meet state health standards for vegetable crops eaten raw, the process train was converted from trickling filters with secondary sedimentation and lagoon sedimentation to activated sludge. A Dissolved Air Floatation/Filtration (DAFF) plant was installed with most importantly, chlorine disinfection. The vegetables produced have gained ready consumer acceptance. A similar reuse project was initiated in 2002 to divert effluent being discharged from Adelaide's Christies Beach WWTP to Gulf St Vincent. The recycled water was sought for a viticulture industry developing on former cereal-growing land. The Willunga Basin Water Company (WBWC) was formed by a founding consortium of growers, winemakers and landholders seeking additional water to supplement limited groundwater for their vines. They initially negotiated access without charge for 10 years to chlorinated Class B/C effluent from the activated-sludge Christies Beach WWTP.

Most of Melbourne's wastewater is treated by its wholesale water provider, Melbourne Water Corporation, at two WWTPs, the Western Treatment Plant (WTP) at Werribee and the Eastern Treatment Plant (ETP) at Bangholme both of which discharge to sea. Both have been upgraded in recent years and support agricultural irrigation. The Werribee WWTP supplements an already established irrigation area adjacent to the Werribee River and the resources are co-managed. Major expansion beyond the original area has since been achieved. It appears that the salinity problems occurred during the Millennium Drought (2000-2011) when urban water use restrictions increased the salinity concentration of the raw effluent entering the WWTP. The Eastern WWTP, which operates with ozonation and biological activated carbon media filtration to produce Class A water for third pipe domestic use by retail provider, Southeast Water, for agricultural use in the Eastern Irrigation Scheme and for ocean discharge. Salinity levels are half those of the recycled water from the Western WWTP.

Water and sewage are the responsibility of local government in most of New South Wales and Queensland. In response to the establishment of environmental discharge standards, some utilities chose to develop their own agricultural irrigation enterprises by buying suitable agricultural land. Crops grown include sugar cane, turf grass and pastures for beef cattle. The WWTPs' agricultural enterprises are managed as separate cost centers, many contractually. In one case, management has been assigned to a local agricultural high school which benefits from the profits. Soil condition and salinity requires careful application management. Owning their own farm gives the WWTP better control of the strategic resource and can better manage inter-seasonal demands for recycled water, Other WWTPs may sell recycled water directly to growers, sometimes investing in the distribution system to the grower's property. However, there have been examples of the demand for recycled water being overestimated. "Take and pay" systems may be in place. Some WWTPs advise of the composition of their recycled water, which may vary seasonally with influent composition, by use of their web sites. Growers can then match the WWTP offering with other water they may be holding.

The Millennium drought (2000-2011) resulted in considerable support for recycled water, but the majority of programs were oriented to urban use. A drought on the east coast of Australia (2017-2020) again brought recycled water to the fore. With the creation of the National Water Grid Authority in 2019, several agricultural irrigation projects were being developed, either supported for construction or for business plan development to allow their long-term economic viability to be assessed.

It is noteworthy that a considerable level of subsidy has been provided for the establishment of most of the recycled water agricultural irrigation systems. Despite water being constitutionally a "states/territories matter", most of the subsidies have come from the Commonwealth (Australian) government. The increasing interest in water recycling for potable use in major cities and towns, although currently practiced only in Perth, may ultimately result in less opportunities for irrigators since there will be greater competition for recycled water to augment drinking water supplies. Further competition may also arise from the use of recycled water as feed-stock for electrolytic hydrogen production as a new fuel source, currently being explored in a pilot plant at a Melbourne WWTP (Radcliffe, 2022).

6.2 California

6.2.1 Introduction

According to the California Department of Water Resources, statewide, average water use is roughly 50% environmental, 40% agricultural, and 10% urban. Currently, recycled water offsets about nine percent of the state's water demand, contributing about 728,000 acre-feet (AF) per year (California's Water Supply Strategy - Adapting to A Hotter, Drier Future, 2022). According to the California's plan to increase water supplies, the state plans to increase recycled water reuse by 0.8 million AF (987 million m^3) per year by 2030 and by 1.8 million AF (2.2 billion m^3) per year in 2040. However, over the last two decades growth in recycled water reuse has been lower than the target as shown in Figure 6-1. This pace might be accelerated by the state's planed investment of more than $27 billion dollars in reuse project to achieve the additional 18 million AF by 2040.

California has a long history of recycled water supply and reuse. From 1970 to 2020, recycled water reuse increased by four-fold from 175,000 AF per Year (216 million m^3 per year) to 728,000 AF per Year (898 million m^3 per year), respectively (Figure 6-2). While recycled water reuse trends have been stagnant in the last decade, there was a modest increase of 0.5% from 728,000 AF of recycled water reuse in 2020 to 732,000 AF (903 million m^3) during the 2021 calendar year probably due to persistent drought conditions. Figure 6-2 shows total recycled water reuse by region in California. Region 8 (Santa Ana) also uses a significant amount of recycled water for potable reuse such as groundwater recharge. The largest recycled water reuse is located in region 4 (Los Angeles) attributed to potable reuse probably due to the large population. Recycled water reuse in major agricultural regions like the Central Valley (region 5) had trended downward over the last two decades. But it's worth noting that the Central Valley has the largest agricultural irrigation water reuse of any region in California. For example, the Turlock and Ceres recycled water projects that supply farmers in westside with recycled water for agricultural irrigation).

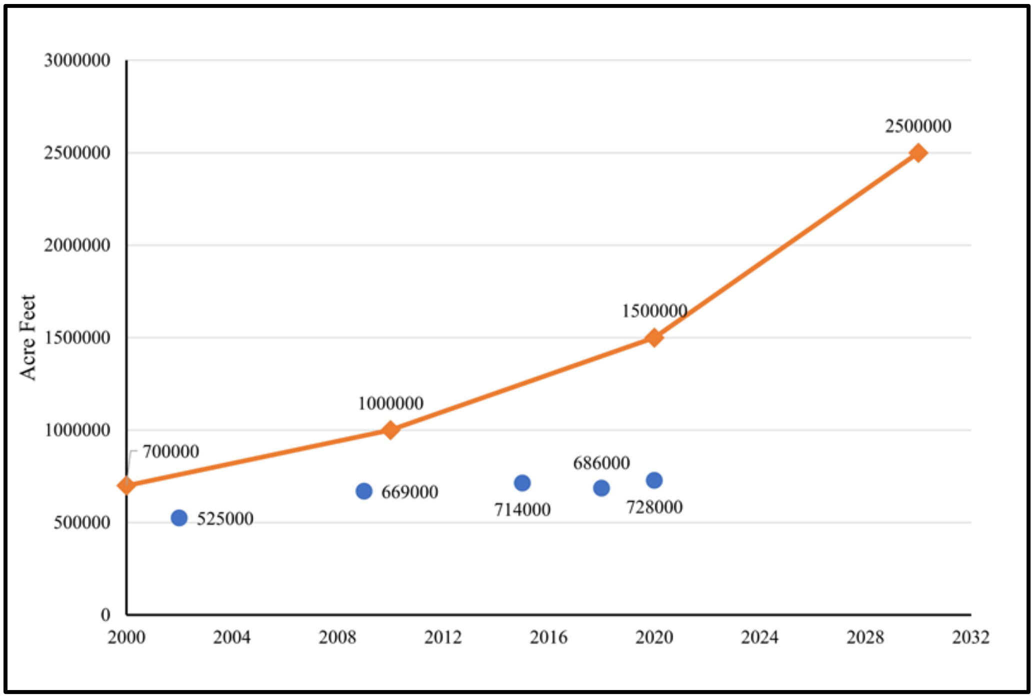

Figure 6-1. California's Recycled Water Reuse Showing Actual Recycled Water Reuse (Blue) Is below Target (Orange).

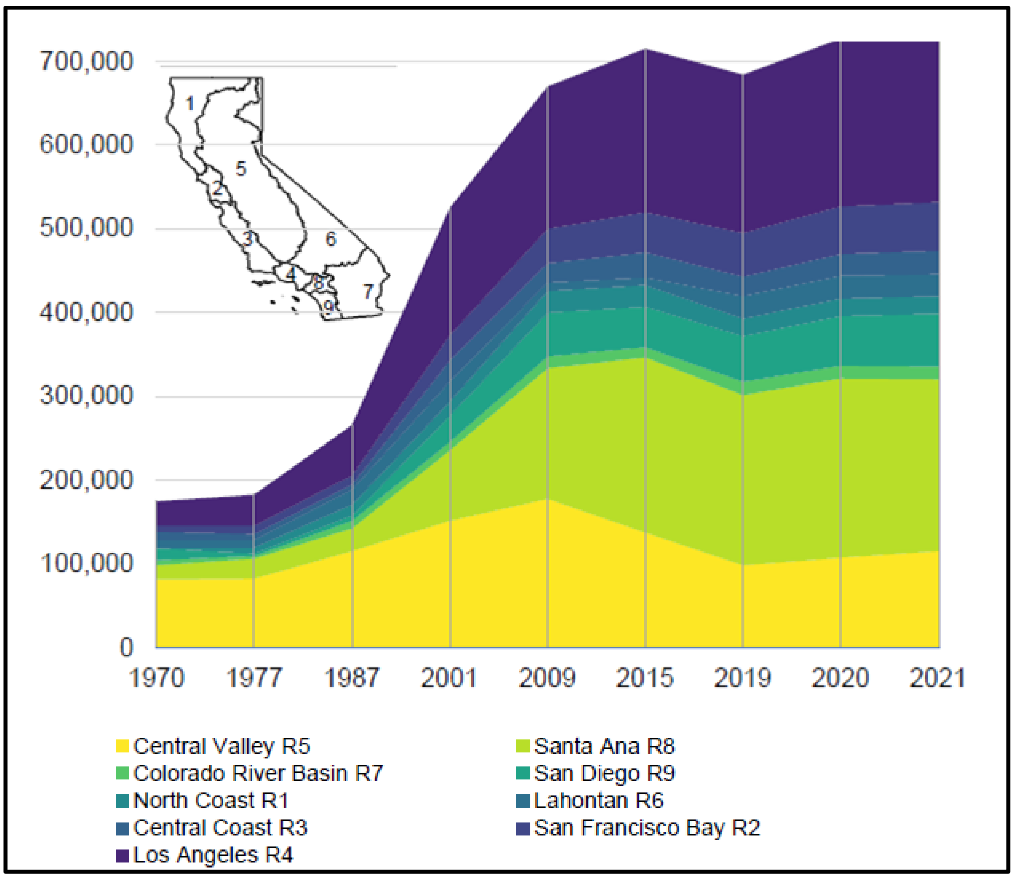

Figure 6-2. Total Recycled Water Reuse (Acre-Feet) by Region in California between 1970-2021.
Source: State Water Resources Control Board.

According to reported recycled water reuse data from 2019 to 2021 (Figure 6-3) by the State Water Board, most recycled water in California is used for landscape irrigation, followed by agricultural irrigation and potable reuse. Other recycled water uses include Industrial and Commercial Applications, Geothermal Energy Production and Other Non-Potable Uses e.g., groundwater recharge and protection from sea water intrusion. It is worth noting that in California if wastewater is not recycled in accordance with Title 22 it cannot be counted as recycled water reuse. For example, if untreated wastewater is discharge on an agricultural field and used to irrigate a crop, that type of use is not considered recycled water reuse. This implies that agricultural reuse of wastewater could be much larger if discharge of wastewater for irrigation was considered a water reuse. Annual recycled water supply and reuse data is now publicly available on a State Water Board dashboard called Geo Tracker.

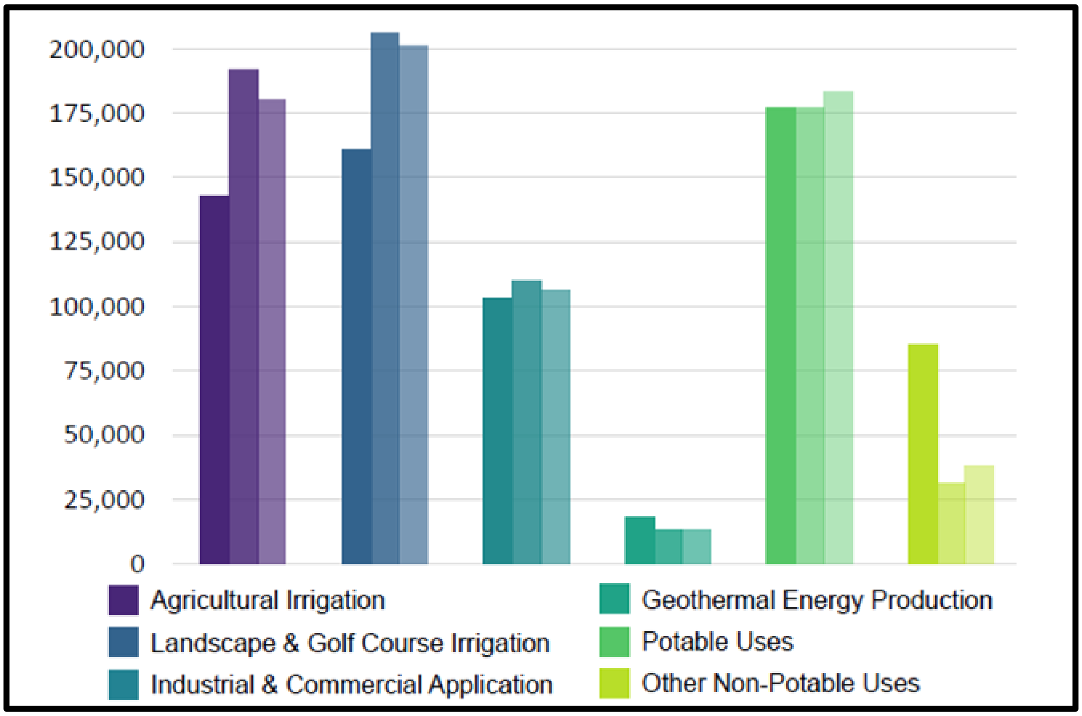

Figure 6-3. Recycled Water Reuse (Acre-Feet) by Category in California between 2019-2021.
Source: State Water Resources Control Board.

6.2.2 California Recycled Water Quality Standards for Irrigation

The primary law that regulates recycled water uses is the California Code of Regulations Title 22. Table 6-1 summarizes regulations for using recycled water in irrigation based on the treatment levels. In general, lower treatment levels have more restrictions on irrigation. Figure 6-4 shows irrigation reuse by treatment level. Almost 100% of uses in landscape and golf course irrigation require disinfected tertiary treatment because of the high probability of public exposure. However, the case is different with agricultural irrigation where about half of uses were irrigated with tertiary recycled water and the other half with undisinfected secondary recycled water.

Table 6-1. Treatment Levels and Irrigation Beneficial Uses Based on the California Code of Regulations Title 22. Uses for Increasing Levels of Treatment Also Include All Uses for Lower Treatment Levels.

Treatment Level	General Description	Beneficial Uses Permitted by Title 22
Advanced	Reverse osmosis, micro-or nanofiltration, ozonation, advanced oxidation	Agriculture, landscape, golf course
Disinfected Tertiary	Oxidized, filtered, and disinfected wastewater to achieve both bacterial and viral removal	Agriculture, landscape, golf course
Disinfected Secondary – 2.2	Oxidized and disinfected wastewater with total coliform bacteria <2.2 MPN/100 ml	Surface irrigated food crops with no direct contact with edible portion by recycled water
Disinfected Secondary – 23	Oxidized and disinfected wastewater with total coliform bacteria <23 MPN/100 ml	Restricted access landscaping and golf courses Sod farms and ornamental nurseries with unrestricted access Pasture for milk animals for human consumption
Undisinfected Secondary	Oxidized wastewater	Orchards with no contact with the edible portion Fodder, pasture and fiber for animals not producing milk and for human consumption. Sod farms, ornamental nursery and nonfood trees not irrigated less than 14 days before harvesting Seed crops not for human consumption Pathogenic-destroyed processed food crops for human consumption

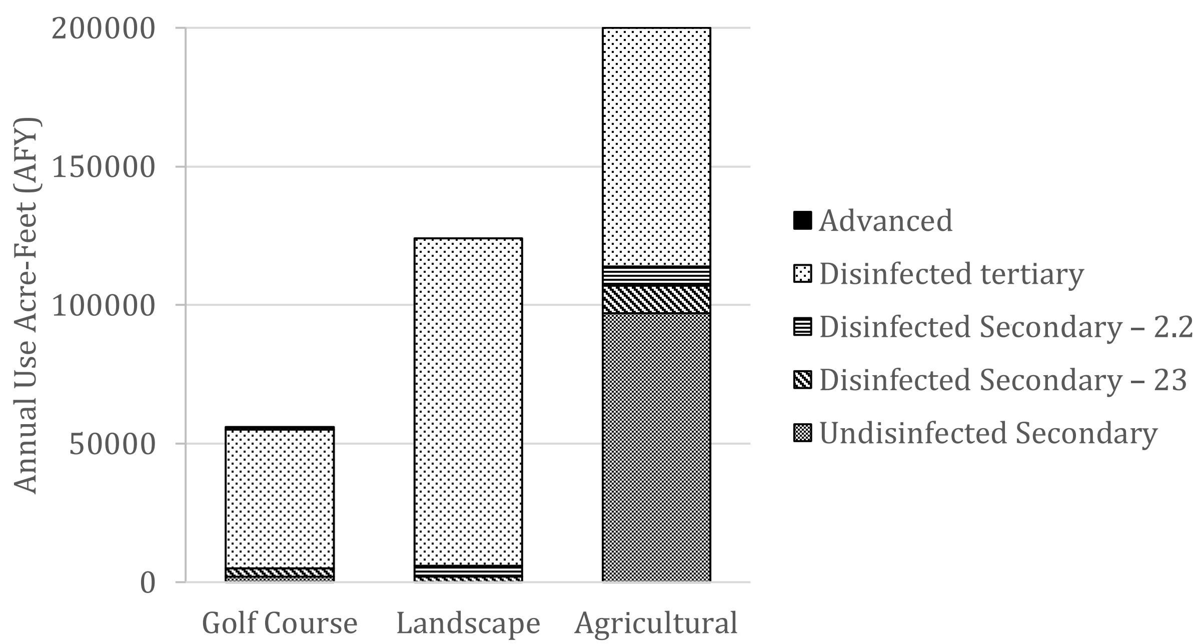

Figure 6-4. Irrigation Beneficial Reuse by Treatment Level.
Source: Pezzetti and Balgobin, 2016.

6.2.3 Public Acceptance of Recycled Water Reuse for Irrigation

Strict regulations and successful case studies helped to build public confidence and acceptance of the practice of reusing recycled water for irrigation. In California the water recycling criteria encoded in Title 22 of the California Code of Administration allows for 43 specified uses of recycled water including irrigation of all types of food crops. These criteria include different water quality requirements for irrigation of each type of crop; those eaten raw, those receiving processing before consumption, and those not involving any human contact before industrial processing (California Agricultural Water Stewardship Initiative, 2022; Sheikh et al., 2019). Title 22 regulations are among the most stringent in the world and have been used as a model for many other countries' guidelines and water reuse regulations. In California, growers using recycled water meeting the Title 22 criteria for example Monterey One and Pajaro Valley Management Agency have shown over the last 50 years (1970s to 2020s) that this practice is safe and sustainable (Olivieri et al., 2014). Recycled water is also sustainable, conserves energy and provides a significant portion of the nutrients needed by the crops nitrogen, phosphorus and micronutrients and enhances water supply resilience especially in the face of climate change. All these factors have combined to increase acceptance of recycled water reuse for irrigation. Another important factor to the success has been education of public engagement. Public agencies such as the regional water quality boards, private entities such as recycled water producers and supplies and organizations such as the California WateReuse have combined efforts to educate the public on the process of purifying wastewater. All the additional disinfection and filtration processes that make it safe for irrigation. As well as strict monitoring required by the local and state Departments of Public Health.

6.2.4 Future Outlook of Recycled Water Reuse in California

California aims to increase recycled water supply and reuse by 2.5 million AF by 2040 but current reported recycled water reuse data is below target goals. However, the State Water Resources Control Board has invested in several recycled water reuse projects since 2015 that are expected to accelerate the use of recycled water in the coming years. Improvements in data reporting through the geo track will allow for annual collection of recycled water supply and reuse data which will facilitate periodic revaluation of available of wastewater influent and recycled water potential. Recycled water reuse is also expected to increase for various beneficial uses including groundwater recharge, landscape irrigation, and agricultural irrigation due to various state funding initiatives.

6.3 Chile

Chile covers an area of 756,102 km^2 and has a vast length of more than 4000 km, bounded on the east by the Andes Mountain Range and on the west by the Pacific Ocean (DGA, 2016; Vera-Puerto et al., 2019). The population of Chile is around 17.5 million, of which 88% is urban and 12% is rural (INE, 2020). The climate of Chile is highly varied and can be categorized into four regional macrozones from north to south: semi-arid and desert (north, 40% of Chilean territory, internationally known as the Atacama Desert), Mediterranean (central), temperate (south), and tundra and glacial (extreme south) (Vera-Puerto et al., 2022).

During the first decade of the 21st century, Chile significantly improved its coverage of wastewater treatment in urban areas, increasing from 20% to almost 90% in 10 years. In 2019, wastewater coverage in Chile was estimated to be near 100%, and the country now has around 300 wastewater treatment plants (WWTPs). In terms of flow, the total wastewater production in urban areas was estimated to be around 40 m^3/s (SISS, 2020). In Chile, typical WWTPs for urban areas are designed mainly for secondary treatment (aerobic technologies represent more than 60%), plus disinfection by chlorination (Vera et al., 2013). In April 2021, 94% of the WWTPs in Chile fulfilled the discharge limits included in the regulations (Vera-Puerto et al., 2022). Therefore, the country has the ability to recycle treated wastewater. However, the reality in rural areas is totally the opposite. It has been estimated by governmental institutions that sewerage coverage is around 25%, while wastewater treatment coverage is less than 10% (MOP, 2020; Mena et al., 2020).

The main objective of the regulatory framework for the sanitary sector was to extend the coverage of sewerage and wastewater treatment to reduce health risks and protect water resources (Vera-Puerto et al., 2022). However, the recycling of treated wastewater was not included in this focus. For the reclamation of treated wastewater for irrigation, Chile, in practice, has followed guideline NCh 1333/87 (INN, 1987), which focuses on different water uses, including irrigation, regardless of the source. Therefore, this guideline is not specific to the reclamation of wastewater. Until now, Chile has not produced a specific regulation focused on the reclamation of wastewater. This lack of specific regulations for the direct recycling of treated wastewater could partially explain the low development in this field, below 0.8% in terms of flow (Villamar et al., 2018; SISS, 2020). Only a few instances in the following activities have been documented regarding the reclamation of treated wastewater: mining, agricultural production, cut flower production, and fodder production (Fundación Chile, 2019; Mena, 2021; Olave et al., 2016).

Historically, Chile has always suffered from drought events. However, since 2010, Chile has been suffering a megadrought that has impacted the central zone (Mediterranean climate), where more than 13 million inhabitants live (79% of the country's population). The longevity of this megadrought is associated with anthropogenic forces, thus showing the influence of climate change on Chile's climate (Garreaud et al., 2019). Given this new scenario, the country needs to find alternative water sources to maintain agricultural production and ensure potable water supplies for its population. The reclamation of treated wastewater alternative mainly for agricultural irrigation is emerging in the discourse in the recent years. The national goal for 2030 is that 30% of wastewater discharge into the sea (at present via marine outfalls), and 20% of recycled wastewater discharged into surface water bodies, will be available for reuse (SISS, 2019). However, to improve the reclamation of treated wastewater in Chile, the institutional framework, regulations, inclusion of rural communities, and emerging compounds, among other aspects must be discussed.

In Chile, water management, including recycled wastewater is a complex issue. At present, more than 40 governmental institutions are involved in water management. Thus, it will be essential to create a unique governmental institution at the national level with the capacity to articulate all the activities related to water (including recycled treated wastewater) if the situation is to improve (Vera-Puerto et al., 2022). In a complementary way, this proposed institution needs guidelines and laws to promote and regulate recycled wastewater use. The challenge lies in enacting these regulations. Until now, Chile has regulated only greywater recycling via Law 21,075, but this law does not include water quality standards (BCN, 2018). Hence, water quality standards must be defined by the Ministry of Health in a new regulation. This new regulation has not yet been promulged, but in 2021, Resolution 404 Exempt was proposed by public consultation (BCN, 2021). In the case of municipal wastewater, the National Institute of Standardization proposed a new guideline package focused on recycled wastewater reuse as irrigation water for agricultural activities: NCh 3456, Parts 1, 2, 3, and 4 (approved on 2021) (INN, 2021). This guideline package includes aspects related to agricultural practices, water quality standards, monitoring, and sampling. In the case of water quality standards, the package includes four categories in Part 2. The four categories while considering the Chilean discharge regulations, and the package is similar to the recent regulation EU 2020/741 (CEC, 2020).

For rural communities, the specific regulatory framework for this sector has recently been updated in Law 20,998 and Decree 50/2020 (BCN, 2017, 2020), which is a general framework not specific to treatment and reuse. Thus, the implementation of treatment and reuse projects with a focus on the circular economy (where reclamation of treated wastewater is a part) will be an important challenge (Vera-Puerto et al., 2022). It is important to modify the present focus of treatment in rural sectors, as more than 70% of decentralized WWTPs (including rural sectors) are based on activated sludge systems without a focus on resource recovery (Vera et al., 2016).

Contaminants of Emerging Concern (CECs), including diclofenac, ibuprofen, naproxen, carbamazepine, fluoxetine, caffeine, sulfamethoxazole, bisphenol A (BPA), atenolol, triclosan, and tonalide, have been reported in effluents of WWTPs in Chile (Reyes-Contreras et al., 2019; Saavedra, 2015). Thus, CECs in Chilean-treated wastewater present a challenge to universities and research centers looking to understand (under local conditions) the effects of CECs on crops and the potential human and animal health risks when treated wastewater is employed as water for irrigation (Vera-Puerto et al., 2022).

Finally, based on the discussed challenges and the scarcity of the national water resources in many populated areas (mainly in the central part of the country), it is expected that the reuse of safe, treated wastewater will increase in the coming years, making recycled treated wastewater a new water source available for irrigation of agricultural crops and to mitigate the current megadrought worsened by climate change.

6.4 Israel

As a semi-arid country, Israel has struggled from its early days to provide a reliable source of water for drinking and agriculture. With intermittent drought years being a common feature of Israel's climate, the expanding agricultural sector was especially affected by fluctuations in water availability for irrigation. Already in the 1950s, in the early days of the State of Israel, officials from the Sanitation Department of the Ministry of Health understood that the best approach to deal with the growing volumes of sewage, is to find allies who will be willing to invest in treated wastewater (TWW) reuse infrastructure. This meant to combine the interests of cities, who could sell TWW to the agricultural sector, thus making them willing to invest in infrastructure, and farmers desperate for a reliable source of water in the face of intermittent droughts and water scarcity (Tal, 2016). Despite TWW's benefits as a reliable source of irrigation water, it also contains potential risks to the environment, soil and public health due to the potentially higher levels of microbial pathogens, salinity, and heavy metals. Over the 75 years of its existence, Israel strived and managed to develop a robust TWW infrastructure and regulatory framework, that lead it to become the largest re-user of TWW per capita in the world.

The first National Master Plan for TWW reuse was proposed already in 1956, with a suggested goal of 150 MCM/yr of TWW reuse for irrigation, more than 10% of the total volume used for irrigation at the time. This ambitious plan was soon put into action, with over 50 related projects implemented by 1962 – linking the TWW from treatment plants to agricultural fields throughout the country (Friedler, 2001; Shaviv et al.; 2011, Tal, 2016). At these earlier stages of TWW reuse, during the 50's and 60's, most treatment was a rudimentary primary treatment, with a lack of stringent requirements regulating TWW quality. While there was little to no evidence for impacts on human health, the high salinity of the TWW was soon found to affect crops and soils negatively (Friedler, 2001; Reznik et al., 2017). Secondary treatment took longer to become a standard practice. The largest wastewater treatment plant in the country (*Shafdan*) was inaugurated in 1969 and was the first plant to utilize *activated sludge* secondary treatment. Over the next few decades, an increasing number of treatment plants began utilizing this technology.

There are many landmarks in the 70-year development of TWW irrigation in Israel. A key landmark is the publication of the '*Shelef Committee Guidelines*' in 1978. These guidelines were first to divide crops into categories (vegetables, fruits, flowers, public lawns) which were each permitted irrigation with TWW of specific quality. The guidelines also designated the category of "TWW for unlimited irrigation", allowing unrestricted irrigation of all crops with tertiary treated TWW (Katz, 2014). A year earlier, the *Shafdan* plant began treating its secondary TWW by 'Soil Aquifer Treatment', i.e., filtering it through the sand dunes adjacent to the plant. The filtered water are pumped out of the aquifer downflow providing a very high-quality TWW for irrigation (Elkayam et al., 2021, Shtull-Trauring et al., 2020). Today, the *Shafdan* provides around third of the total TWW used for irrigation (200 MCM/yr, Water Authority, 2020).

The first officially codified statutory TWW regulation were adopted in 1981, requiring farmers that irrigate with TWW to get annual permits, and prohibiting irrigation with untreated wastewater (Ministry of Health, 1981), and in 1992, a standard requiring 20 ppm for BOD and 30 ppm for TSS was adopted (Ministry of Health, 1992). At this point, Israel had already surpassed the 1956 Masterplan goal and was reusing ~160 MCM/yr TWW for irrigation, which amounts to 13% of the total agricultural irrigation water (Central Bureau of Statistics, 2020). However, with volumes of produced TWW increasing with the steady increase in population, it soon became clear that these basic standards are insufficient to deal with damage to crops and soils due to irrigation with large volumes of low quality TWW and the risk to public health (Tal, 2016).

Beginning in the early 90's many more TWW reservoirs were being constructed. In the next 20 years, over 100 reservoirs were built with a capacity of over 200 MCM/yr (Cohen and Harel, 2010). Another major landmark in TWW history in Israel was the publication of the '*Halperin Committee report*' in 1999. The stated aim of the committee was to modernize the earlier regulations for modern agricultural irrigation practices. One important innovation was adopting a strict microbial quality standard. Despite of recommendations at the time to adopt 100 CFU/100 mL as the threshold level as 50 year of experience showed no apparent risk due to exposure to TWW, the committee adopted a stricter standard of 10 CFU/100 mL with the declared goal to avoid even the slightest risk of morbidity outbreaks and facilitate export to all international markets. This exemplifies the forward thinking of Israel's public health and water regulators.

Another important innovation of the committee was the development of the "Barrier" system for TWW irrigation. Under this system, agronomic practices that minimize exposure of the plant product to the TWW, and TWW practices that improve the TWW microbial quality are considered 'Barriers' for contamination. Depending on the TWW quality or the sensitivity of the crop category for transmittance of bacterial human pathogens to the consumer, the guidelines require a varying number of "Barriers" to be applied. For instance, holding the treated water 30 days in a closed TWW reservoir counts as one "barrier", cultivation with plastic soil mulch counts as one "barrier", while sub-surface drip irrigation counts as two "barriers" (Aloni et al., 1999). These "Barriers" thus served as an incentive for the adoption of preventive cultivation practices by farmers and for the construction of TWW reservoirs. One year after the '*Halperin committee guidelines*' were published, in 2000, Israel was already reusing 260 MCM/yr TWW for irrigation, ~22% of total irrigation water, that represents an increase of over 60% in TWW reuse for irrigation in a decade (Water Authority, 2020).

Another important regulation that was mandated in 2000, is that all industrial wastewater exceeding specific standards, will undergo pre-treatment before being released into the municipal waste system; with the stated goal of "Protecting water sources from metals and other contaminants" (Ministry of Environmental Protection, 2000). Around the same time, between 1998-2002, successive years of drought severely affected availability of water for irrigation in Israel. Besides driving the decision to build large-scale desalination plants for drinking water, the drought crisis also forced the agricultural sector to cut its freshwater use by almost half – from ~920 million m^3/yr in 1998 to ~530/ million m^3 in 2002. While initially, this led to an overall decrease in total water use for irrigation, TWW quickly began to fill the gap. Ever since, freshwater irrigation continued to decrease until stabilizing at around an average of ~450 MCM/yr, and TWW volumes steadily increased. In 2011, for the first time more TWW (~415 MCM/yr) was being used for irrigation than freshwater (~414 MCM/yr) (Water Authority, 2020).

The modern era of TWW irrigation standards in Israel began in 2010, when the TWW Public Health Standards, also known as the "*Inbar Committee*" Standards were signed into law (Ministry of Health, 2010). There are at least three important innovations in these standards. First, they include a wide range (36) parameters for water quality. Second, they include a standard of water quality allowed for release into streams. Third, it mandated a thorough monitoring scheme of TWW quality in plants and reservoirs (Inbar, 2010; Ministry of Health, 2010). This detailed monitoring scheme provides the opportunity to use the collected data to analyze spatial and temporal trends in TWW quality. This has allowed to evaluate the significant scale of reduction in TWW salinity due to large-scale desalination (Shtull-Trauring et al., 2020), and to assess the sustainability value of "free" nutrients in TWW for crop fertilization (Shtull-Trauring et al., 2022).

In 2010, it was stated that "The objective is to treat 100% of the country's wastewater to a level enabling unrestricted irrigation in accordance with soil sensitivity and without risk to soil and water sources" (Inbar, 2010). While the 100% treatment and reuse were not achieved yet, Israel has reached an impressive scale of TWW irrigation infrastructure, a result of the ambitious plans set out by the water regulators of the country over decades. By 2020, a decade later, Israel collects almost 95% of its raw sewage for treatment in secondary (or tertiary) plants. 85% of the TWW (570 MCM/yr) is reused for irrigation, amounting to about half the total irrigation water volume (freshwater provided only 35%) (Figure 6-5) (Cohen et al., 2020; Water Authority, 2020). This does not mean there are no issues requiring attention. 40% of the TWW is still receiving only secondary treatment and ~55% of the treatment plants are operating over 90% capacity, with at least 13 treatment plants working at 100% capacity (Ministry of Environmental Protection, 2021). This means an unavoidable occasional discharge of low quality TWW overflows into rivers. There is still a small percentage of raw sewage that is released to rivers (some by permit, some due to failure of the treatment infrastructure).

To achieve Inbar's goal of 100% TWW for unrestricted irrigation, further investment is required in Israel's TWW infrastructure. Nonetheless, Israel has managed to show that achieving this goal is feasible. In the last 70 years, the country invested in and developed an extensive treatment infrastructure – plants, reservoirs and pipelines - as well as a robust regulatory framework and monitoring scheme to help maximize reuse of TWW for irrigation while minimizing the risk to public health, agriculture and the environment. As water scarcity becomes an increasingly global problem, especially in semi-arid regions, the lessons learned from Israel's experience with TWW can help the many countries who are still not utilizing the huge potential of TWW irrigation – turning sewage from a public health and environmental hazard into a valuable resource, nutrient rich irrigation water.

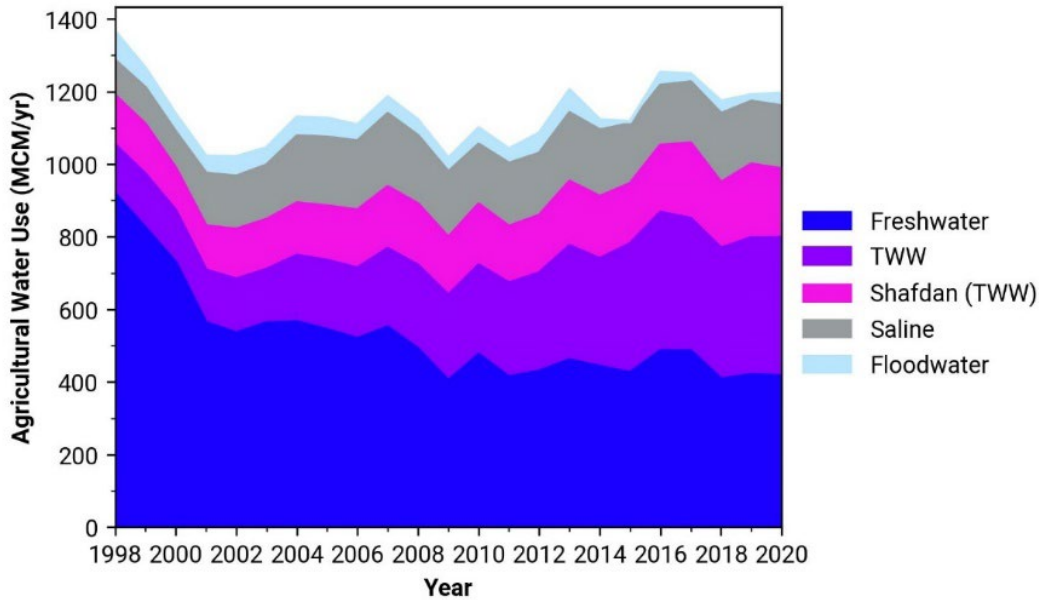

Figure. 6-5. Agricultural Water Use in Israel by Water Type.
Source: Generated from data from the Water Authority, 2020.

6.5 Spain

6.5.1 Historical Background Water Reuse for Irrigation in Spain

Many Countries such as Spain, facing water scarcity have long been aware of the potential of technologies for combatting water shortages. In Spain, which suffers from chronic structural water deficits, water reuse and desalination have been a priority for some time now. Spain has a highly variable rainfall regime, averaging over 2000 mm/y in some areas (Galicia, the Cantabrian Mountains, the Basque-Navarran Pyrenees, the Central Mountain System and the Sierra de Ubrique) but less than 200 mm/y in the southeast (Almeria and Murcia), one of the lowest rainfall levels in Europe. This complicated water balance is becoming particularly acute in some areas of the country; the Mediterranean coast, which already suffers from water scarcity, is the region worst hit by droughts. Practically all surface water resources in Spain are already stored in reservoirs, and so no new reservoirs are scheduled for construction in the near future. Furthermore, in many cases, groundwater resources are overexploited. In view of the circumstances described above, no significant future increases are expected in available water from conventional sources, and so in the most vulnerable areas a key role will be played by alternative sources such as recycled water and desalination of brackish water and seawater.

In Spain, the actions taken in matters concerning sewerage and treatment began in the 1970s with the implementation of partial plans in some tourist areas on the coast, the greatest investment drive has taken place over the last decade, after the Directive 91/271/EEC came into force and was applied to Spanish Law via Act 11/1995 and Royal Decree 509/1996 (LBA,2000). With a view to ensuring compliance with the new legislation, the Central Government, through the Ministry of Environment and with the collaboration of the Autonomous Regions, drew up the National Sewerage and Treatment Plan (PNSD) as an essential part of the planning process for the different infrastructures that had to be provided in Spain before 2005 in the area of sewerage and treatment, and this became the tool that coordinated the different Regional Administrations with powers in the matter. In 2001, there were approximately 140 reuse activities to cover a demand of around 346 hm3 /y (Catalinas and Ortega, 2002), whereas in 2004, this figure had risen to 408 hm3/y (Iglesias, 2005). Table 6-2 compares the volumes of water reused in 2001 and 2004 and shows how the different types of use have evolved. As can be seen from this table, irrigation is the most widespread use in both periods, although a growing trend towards uses with environmental aims has also been detected.

Table 6-2. Recycled Water Reuse in Spain on the Basis of the Uses in the Years 2001 and 2004.
Source: INE 2020.

Uses	Year 2001		Year 2004	
	Volume (h^3 /y)	%	Volume (h^3 /y)	%
Irrigation	284.9	82.3	323.0	79.2
Municipal Uses	24.0	7.0	33.0	8.1
Recreational use and golf courses	20.6	6.0	25.0	6.0
Industrial Uses	2.5	0.7	3.0	0.7
Ecological uses	14.0	4.0	24.0	6.0
Total	346.0	100.0	408.0	100.0

According to the available data for year 2016, more than 60% of recycled water for reuse in Spain was destined for agriculture. Being the biggest consumer of water in Spain, it is natural that agricultural activity is the most impacted by water scarcity and the most interested in alternative sources. Gardens, leisure, and sport area irrigation (21%) is the second largest consumer, mostly represented by irrigation of public parks and golf courses. Industrial use represents only 5% of the total, while street cleaning represents a tiny proportion, restricted to big cities (Figure 6-6).

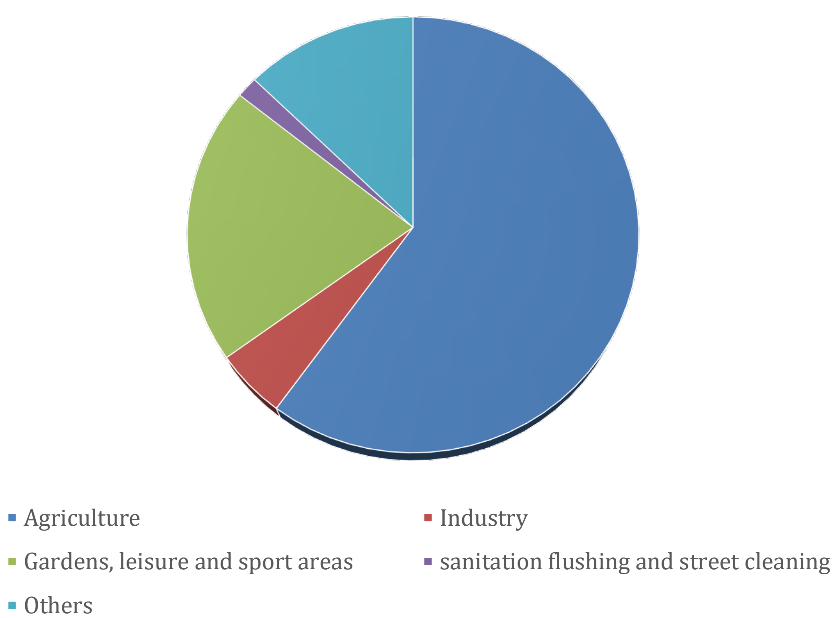

Figure 6-6. Recycled Water Reuse by Category in Spain in 2016.
Source: INE (2020).

Spain stands out as the country with the highest yearly reuse volume of the European Union, with a quantity that exceeds 300 hm^3/year. Albeit a high volume, it lacks behind the expectations set in the 2012 National Plan for Water Reuse, which set an objective above 1000 hm^3/year for 2020 (Navarro, 2018). The accurate quantities of reused volumes remain elusive, since different administrations provide different volumes: infrastructure capacity, wastewater treated to reuse quality standards, treated water reused, etc. The spatial distribution of reused water in Spain is represented respectively in the following Table 6-3 and Figure 6-7 (INE and AEAS, 2020).

Table 6-3. Zones in Spain Where Recycled Water Reuse Is Significant.

Zones	Volumes Reused (hm^3/y)
Communidad Valenciana	128.0
Communidad de Murcia	106.0
Islas Canarias	47.5
Islas Baleares	40.0
Cataluña	33.0
Costa mediterránea Andaluza	11.5
Vitoria-Gaztei	6.5
Madrid	5.0

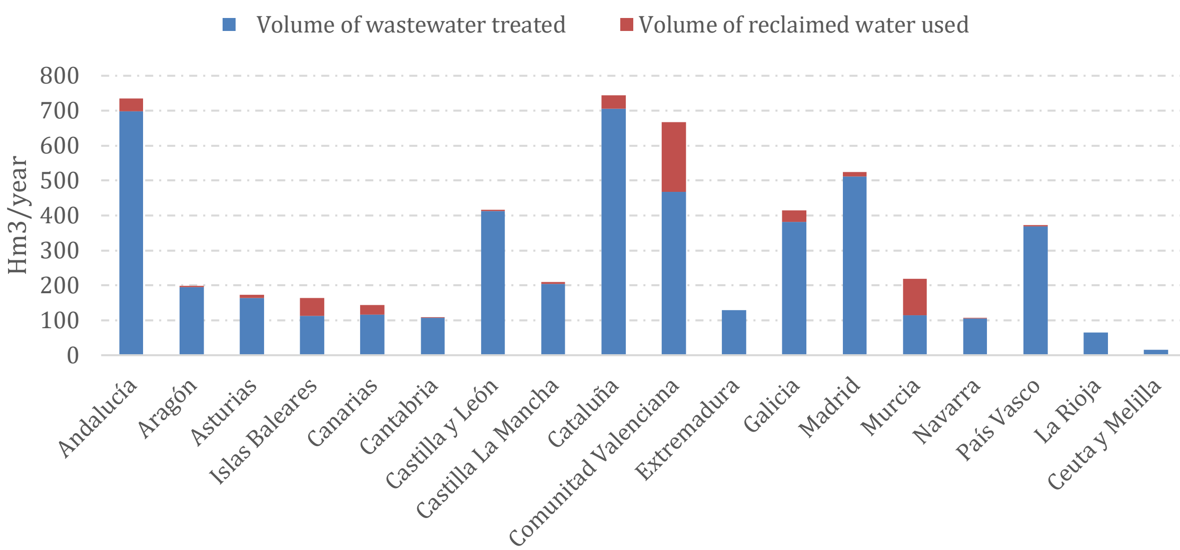

Figure 6-7. Volume of Treated Wastewater in Spain Autonomous Communities In 2016.
Source: INE (2020).

6.5.2 Current Progress on Water Reuse for Irrigation in Spain

Spain leads the European reuse with almost half of the total volume and it is ranked fifth in the world in terms of installed capacity. 27% of the 2,000 WWTPs have tertiary treatment including large plants with advanced technologies (membranes, advanced oxidation and disinfection, etc.). Distribution of reuse within Spain is very uneven. More than 80% of the total is concentrated in the Valencian Community, Murcia, Andalusia, Canary Islands and Balearic Islands (the areas with greater water stress and important agricultural activity), with Murcia Region representing the higher reuse rate, close to 90% of treated wastewater and agriculture irrigation as the main consumer (49% in 2020). From the point of view of the river basins, the contribution of the Jucar and Segura River Basins (SE of Spain, Mediterranean coast) represents about 60% of the whole reuse in Spain.

Also, in the main water reuse Region in Spain (Murcia), they are developing different experimental strategies on water reuse for agriculture at different levels (plot and district working with the irrigation communities). The main objective of that research was to assess the success of multidisciplinary approaches for recycled water use projects in agriculture laying the foundations of novel and more efficient crop production management practices by enabling the saline reclaimed water to be used for irrigation. To achieve this aim, the following methodology was implemented: evaluate the horticultural crops performance, determine the water use efficiency (WUE), establish new agronomic thresholds, develop plant uptake models to evaluate the short and long-term effects of recycled water irrigation and assess practices under Mediterranean field conditions. Finally, the integration of new tools such as GIS or remote sensing enhanced public perception into the water resource use studies (Alcón, et al., 2012; Allende et al., 2016; Nicolás et al., 2012; Pedrero et al., 2009, ,2013,2015a,2015b; Romero-Trigueros et al., 2019; Pedrero Salcedo, F. et al., 2022).

One of the greatest challenges to the global implementation of recycled water reuse is regulations, which not only have great disparity throughout the world but are also practically non-existent in many countries, representing an important barrier to the use of recycled water and other related economic activities such as agriculture. Spain incorporated reuse regulation to the law by means of the Royal Decree 1620/2007, which has been an important tool to develop and to order the application of reclaimed water to different uses (14 uses grouped into 5 categories: urban, agriculture, recreational and environmental uses, with different water quality requirements) and setting the procedures for authorizations, concessions, control, etc. This has been the legal framework in effect for all the reuse activities in Spain until the past year, when European Union launched the Regulation (EU) 2020/741 on minimum requirements for water reuse. This Regulation applies only to the reuse for agriculture (by establishing 4 different water qualities for different conditions) and leaves the rest of uses (industrial, environmental, etc.) up to the Member States.

6.5.2.1 Future Outlook

After the pandemic and economic crisis, a hopeful future opens up over the potential of recycled water reuse growing thanks to the different financial instruments which have been recently launched and can be applied to reuse:

- The so-called “Next Generation” European funds for reconstruction (approximately EUR 140,000 million for Spain for the period 2021-2026) where one of its pillars is the Ecological Transition.
- The Spanish plan DSEAR (acronym for wastewater treatment, sanitation, efficiency, saving and reuse) endowed with EUR 10,000 million for the next 18 years, and
- The European R&D program called HORIZON EUROPE, with an investment of EUR 100,000 million for the period 2021-2027, whose objectives are the fight against climate change, the

contribution to the UN Sustainable Development Goals and boosting the Union's competitiveness and growth.

However, its implementation also brings a series of challenges that need to be considered. A negative social perception is considered as one of the biggest difficulties for the success of a recycled water reuse project. The main factors identified are: disgust generated by the reclaimed water origin, health risk concerns regarding the consumption of crops irrigated with reclaimed water, distrust of the authorities managing the water sources, and preference for conventional water sources. Therefore, to promote the use of reclaimed water in agriculture it is essential to involve the target community from the planning phase of the project and guarantee their continuous participation during the whole process.

Finally, although recycled water is commonly and successfully used in many countries, water reuse face numerous barriers. Therefore, for the preservation of profitable intensive agriculture that protects the environment, innovative agricultural projects incorporating state of the art technologies for sustainable recycled water reuse are needed.

APPENDIX A

A.1 Salt Tolerance of Herbaceous Crops.†

Source: Adapted from Maas and Grattan (1999).

Crop			Salt Tolerance Parameters		
Common Name	**Botanical Name**‡	**Tolerance Based on:**	**Threshold**§ **(EC_e)**	**Slope**	**Rating**¶
			dS/m	% per dS/m	
Fiber, grain and special crops					
Artichoke, Jerusalem	*Helianthus tuberosus* L.	Tuber yield	0.4	9.6	MS
Barley[#]	*Hordeum vulgare* L.	Grain yield	8.0	5.0	T
Canola or rapeseed	*Brassica campestris* L. [syn. *B. rapa* L.]	Seed yield	9.7	14	T
Canola or rapeseed	*B. napus* L.	Seed yield	11.0	13	T
Chickpea	*Cicer arietinum* L.	Seed yield	--	--	MS
Corn[‡‡]	*Zea mays* L.	Ear FW	1.7	12	MS
Cotton	*Gossypium hirsutum* L.	Seed cotton yield	7.7	5.2	T
Crambe	*Crambe abyssinica* Hochst. ex R.E. Fries	Seed yield	2.0	6.5	MS
Flax	*Linum usitatissimum* L.	Seed yield	1.7	12	MS
Guar	*Cyamopsis tetragonoloba* (L). Taub.	Seed yield	8.8	17	T
Kenaf	*Hibiscus cannabinus* L.	Stem DW	8.1	11.6	T
Lesquerella	*Lesquerella fenderli* (Gray) S. Wats.	Seed yield	6.1	19	MT
Millet, channel	*Echinochloa turnerana* (Domin) J.M. Black	Grain yield	--	--	T
Oats	*Avena sativa* L.	Grain yield	--	--	T
Peanut	*Arachis hypogaea* L.	Seed yield	3.2	29	MS
Rice, paddy	*Oryza sativa* L.	Grain yield	3.0[§§]	12[§§]	S
Roselle	*Hibiscus sabdariffa* L.	Stem DW	--	--	MT
Rye	*Secale cereale* L.	Grain yield	11.4	10.8	T
Safflower	*Carthamus tinctorius* L.	Seed yield	--	--	MT
Sesame[¶¶]	*Sesamum indicum* L.	Pod DW	--	--	S
Sorghum	*Sorghum bicolor* (L.) Moench	Grain yield	6.8	16	MT
Soybean	*Glycine max* (L.) Merrill	Seed yield	5.0	20	MT
Sugarbeet[##]	*Beta vulgaris* L.	Storage root	7.0	5.9	T
Sugarcane	*Saccharum officinarum* L.	Shoot DW	1.7	5.9	MS
Sunflower	*Helianthus annuus* L.	Seed yield	4.8	5.0	MT
Triticale	*X Triticosecale* Wittmack	Grain yield	6.1	2.5	T
Wheat	*Triticum aestivum* L.	Grain yield	6.0	7.1	MT
Wheat (semidwarf)[†††]	*T. aestivum* L.	Grain yield	8.6	3.0	T
Wheat, Durum	*T. turgidum* L. var. *durum* Desf.	Grain yield	5.9	3.8	T
Grasses and forage crops					
Alfalfa	*Medicago sativa* L.	Shoot DW	2.0	7.3	MS

Crop			Salt Tolerance Parameters		
Common Name	Botanical Name‡	Tolerance Based on:	Threshold§ (EC_e)	Slope	Rating¶
Alkaligrass, Nuttall	*Puccinellia airoides* (Nutt.) Wats. & Coult.	Shoot DW	--	--	T*
Alkali sacaton	*Sporobolus airoides* Torr.	Shoot DW	--	--	T*
Barley (forage)#	*Hordeum vulgare* L.	Shoot DW	6.0	7.1	MT
Bentgrass, creeping	*Agrostis stolonifera* L.	Shoot DW	--	--	MS
Bermudagrass‡‡‡	*Cynodon dactylon* (L.) Pers.	Shoot DW	6.9	6.4	T
Bluestem, Angleton	*Dichanthium aristatum* (Poir.) C.E. Hubb. [syn. *Andropogon nodosus* (Willem.) Nash]	Shoot DW	--	--	MS*
Broadbean	*Vicia faba* L.	Shoot DW	1.6	9.6	MS
Brome, mountain	*Bromus marginatus* Nees ex Steud.	Shoot DW	--	--	MT*
Brome, smooth	*B. inermis* Leyss	Shoot DW	--	--	MT
Buffelgrass	*Pennisetum ciliare* (L). Link. [syn. *Cenchrus ciliaris*]	Shoot DW	--	--	MS*
Burnet	*Poterium sanguisorba* L.	Shoot DW	--	--	MS*
Canarygrass, reed	*Phalaris arundinacea* L.	Shoot DW	--	--	MT
Clover, alsike	*Trifolium hybridum* L.	Shoot DW	1.5	12	MS
Clover, Berseem	*T. alexandrinum* L.	Shoot DW	1.5	5.7	MS
Clover, Hubam	*Melilotus alba* Dest. var. *annua* H.S.Coe	Shoot DW	--	--	MT*
Clover, ladino	*Trifolium repens* L.	Shoot DW	1.5	12	MS
Clover, Persian	*T. resupinatum* L.	Shoot DW	--	--	MS*
Clover, red	*T. pratense* L.	Shoot DW	1.5	12	MS
Clover, strawberry	*T. fragiferum* L.	Shoot DW	1.5	12	MS
Clover, sweet	*Melilotus sp*. Mill.	Shoot DW	--	--	MT*
Clover, white Dutch	*Trifolium repens* L.	Shoot DW	--	--	MS*
Corn (forage)††	*Zea mays* L.	Shoot DW	1.8	7.4	MS
Cowpea (forage)	*Vigna unguiculata* (L.) Walp.	Shoot DW	2.5	11	MS
Dallisgrass	*Paspalum dilatatum* Poir.	Shoot DW	--	--	MS*
Dhaincha	*Sesbania bispinosa* (Linn.) W.F. Wight [syn. *Sesbania aculeata* (Willd.) Poir]	Shoot DW	--	--	MT
Fescue, tall	*Festuca elatior* L.	Shoot DW	3.9	5.3	MT
Fescue, meadow	*Festuca pratensis* Huds.	Shoot DW	--	--	MT*
Foxtail, meadow	*Alopecurus pratensis* L.	Shoot DW	1.5	9.6	MS
Glycine	*Neonotonia wightii* [syn. *Glycine wightii* or *javanica*]	Shoot DW	--	--	MS
Gram, black or Urd bean	*Vigna mungo* (L.) Hepper [syn. *Phaseolus mungo* L.]	Shoot DW	--	--	S
Grama, blue	*Bouteloua gracilis* (HBK) Lag. ex Steud.	Shoot DW	--	--	MS*
Guinea grass	*Panicum maximum* Jacq.	Shoot DW	--	--	MT
Hardinggrass	*Phalaris tuberosa* L. var. *stenoptera* (Hack) A. S. Hitchc.	Shoot DW	4.6	7.6	MT
Kallargrass	*Leptochloa fusca* (L.) Kunth [syn. *Diplachne fusca* Beauv.]	Shoot DW	--	--	T
Kikuyugrass	*Pennisetum clandestinum* L.	Shoot DW	8.0		T

Crop			Salt Tolerance Parameters		
Common Name	Botanical Name‡	Tolerance Based on:	Threshold§ (EC_e)	Slope	Rating¶
Lablab bean	*Lablab purpureus* (L.) Sweet [syn. *Dolichos lablab* L.]	Shoot DW	--	--	MS
Lovegrass§§§	*Eragrostis sp.* N. M. Wolf	Shoot DW	2.0	8.4	MS
Milkvetch, Cicer	*Astragalus cicer* L.	Shoot DW	--	--	MS*
Millet, Foxtail	*Setaria italica* (L.) Beauvois	Dry matter	--	--	MS
Oatgrass, tall	*Arrhenatherum elatius* (L.) Beauvois ex J. Presl & K. Presl	Shoot DW	--	--	MS*
Oats (forage)	*Avena sativa* L.	Straw DW	--	--	T
Orchardgrass	*Dactylis glomerata* L.	Shoot DW	1.5	6.2	MS
Panicgrass, blue	*Panicum antidotale* Retz.	Shoot DW	--	--	MS*
Pigeon pea	*Cajanus cajan* (L.) Huth [syn. *C. indicus* (K.) Spreng.]	Shoot DW	--	--	S
Rape (forage)	*Brassica napus* L.		--	--	MT*
Rescuegrass	*Bromus unioloides* HBK	Shoot DW	--	--	MT*
Rhodesgrass	*Chloris Gayana* Kunth.	Shoot DW	--	--	MT
Rye (forage)	*Secale cereale* L.	Shoot DW	7.6	4.9	T
Ryegrass, Italian	*Lolium multiflorum* Lam.	Shoot DW	--	--	MT*
Ryegrass, perennial	*Lolium perenne* L.	Shoot DW	5.6	7.6	MT
Ryegrass, Wimmera	*L. rigidum* Gaud.		--	--	MT*
Saltgrass, desert	*Distichlis spicta* L. var. *stricta* (Torr.) Bettle	Shoot DW	--	--	T*
Sesbania	*Sesbania exaltata* (Raf.) V.L. Cory	Shoot DW	2.3	7.0	MS
Sirato	*Macroptilium atropurpureum* (DC.) Urb.	Shoot DW	--	--	MS
Sphaerophysa	*Sphaerophysa salsula* (Pall.) DC	Shoot DW	2.2	7.0	MS
Sudangrass	*Sorghum sudanense* (Piper) Stapf	Shoot DW	2.8	4.3	MT
Timothy	*Phleum pratense* L.	Shoot DW	--	--	MS*
Trefoil, big	*Lotus pedunculatus* Cav.	Shoot DW	2.3	19	MS
Trefoil, narrowleaf birdsfoot	*L. corniculatus* var *tenuifolium* L.	Shoot DW	5.0	10	MT
Trefoil, broadleaf birdsfoot	*L. corniculatus* L. var *arvenis* (Schkuhr) Ser. ex DC	Shoot DW	--	--	MS
Vetch, common	*Vicia angustifolia* L.	Shoot DW	3.0	11	MS
Wheat (forage)†††	*Triticum aestivum* L.	Shoot DW	4.5	2.6	MT
Wheat, Durum (forage)	*T. turgidum* L. var *durum* Desf.	Shoot DW	2.1	2.5	MT
Wheatgrass, standard crested	*Agropyron sibiricum* (Willd.) Beauvois	Shoot DW	3.5	4.0	MT
Wheatgrass, fairway crested	*A. cristatum* (L.) Gaertn.	Shoot DW	7.5	6.9	T
Wheatgrass, intermediate	*A. intermedium* (Host) Beauvois	Shoot DW	--	--	MT*
Wheatgrass, slender	*A. trachycaulum* (Link) Malte	Shoot DW	--	--	MT

Crop			Salt Tolerance Parameters		
Common Name	Botanical Name[‡]	Tolerance Based on:	Threshold[§] (EC_e)	Slope	Rating[¶]
Wheatgrass, tall	*A. elongatum* (Hort) Beauvois	Shoot DW	7.5	4.2	T
Wheatgrass, western	*A. smithii* Rydb.	Shoot DW	--	--	MT*
Wildrye, Altai	*Elymus angustus* Trin.	Shoot DW	--	--	T
Wildrye, beardless	*E. triticoides* Buckl.	Shoot DW	2.7	6.0	MT
Wildrye, Canadian	*E. canadensis* L.	Shoot DW	--	--	MT*
Wildrye, Russian	*E. junceus* Fisch.	Shoot DW	--	--	T
Vegetable and fruit crops					
Artichoke	*Cynara scolymus* L.	Bud yield	6.1	11.5	MT
Asparagus	*Asparagus officinalis* L.	Spear yield	4.1	2.0	T
Bean, common	*Phaseolus vulgaris* L.	Seed yield	1.0	19	S
Bean, lima	*P. lunatus* L.	Seed yield	--	--	MT*
Bean, mung	*Vigna radiata* (L.) R. Wilcz.	Seed yield	1.8	20.7	S
Cassava	*Manihot esculenta* Crantz	Tuber yield	--	--	MS
Beet, red[##]	*Beta vulgaris* L.	Storage root	4.0	9.0	MT
Broccoli	*Brassica oleracea* L. (Botrytis Group)	Head FW	1.3	15.8	MT
Brussels Sprouts	*B. oleracea L.* (Gemmifera Group)		--	--	MS*
Cabbage	*B. oleracea* L. (Capitata Group)	Head FW	1.8	9.7	MS
Carrot	*Daucus carota* L.	Storage root	1.0	14	S
Cauliflower	*Brassica oleracea* L. (Botrytis Group)		1.5	14.4	MS*
Celery	*Apium graveolens* L. var *dulce* (Mill.) Pers.	Petiole FW	1.8	6.2	MT
Corn, sweet	*Zea mays* L.	Ear FW	1.7	12	MS
Cowpea	*Vigna unguiculata* (L.) Walp.	Seed yield	4.9	12	MT
Cucumber	*Cucumis sativus* L.	Fruit yield	2.5	13	MS
Eggplant	*Solanum melongena* L. var *esculentum* Nees.	Fruit yield	1.1	6.9	MS
Fennel	*Foeniculum vulgare* Mill.	Bulb yield	1.4	16	S
Garlic	*Allium sativum* L.	Bulb yield	3.9	14.3	MS
Gram, black or Urd bean	*Vigna mungo* (L.) Hepper [syn. *Phaseolus mungo* L.]	Shoot DW	--	--	S
Kale	*Brassica oleracea* L. (Acephala Group)		--	--	MS*
Kohlrabi	*Brassica oleracea* L. (Gongylodes Group)		--	--	MS*
Lettuce	*Lactuca sativa* L.	Top FW	1.3	13	MS
Muskmelon	*Cucumis melo* L. (Reticulatus Group)	Fruit yield	1.0	8.4	MS
Okra	*Abelmoschus esculentus* (L.) Moench	Pod yield	--	--	MS
Onion (bulb)	*Allium cepa* L.	Bulb yield	1.2	16	S
Onion (seed)		Seed yield	1.0	8.0	MS
Parsnip	*Pastinaca sativa* L.		--	--	S*

Crop		Tolerance Based on:	Salt Tolerance Parameters		
Common Name	Botanical Name‡		Threshold§ (EC_e)	Slope	Rating¶
Pea	*Pisum sativum* L.	Seed FW	3.4	10.6	MS
Pepper	*Capsicum annuum* L.	Fruit yield	1.5	14	MS
Pigeon pea	*Cajanus cajan* (L.) Huth [syn. *C. indicus* (K.) Spreng.]	Shoot DW	--	--	S
Potato	*Solanum tuberosum* L.	Tuber yield	1.7	12	MS
Pumpkin	*Cucurbita pepo* L. var *Pepo*		--	--	MS*
Purslane	*Portulaca oleracea* L.	Shoot FW	6.3	9.6	MT
Radish	*Raphanus sativus* L.	Storage root	1.2	13	MS
Spinach	*Spinacia oleracea* L.	Top FW	2.0	7.6	MS
Squash, scallop	*Cucurbita pepo* L. var *melopepo* (L.) Alef.	Fruit yield	3.2	16	MS
Squash, zucchini	*C. pepo* L. var *melopepo* (L.) Alef.	Fruit yield	4.9	10.5	MT
Strawberry	*Fragaria* x *Ananassa* Duch.	Fruit yield	1.0	33	S
Sweet potato	*Ipomoea batatas* (L.) Lam.	Fleshy root	1.5	11	MS
Swiss chard	*Beta vulgaris* L.	Top FW	7.0	5.7	T
Tepary bean	*Phaseolus acutifolius* Gray		--	--	MS*
Tomato	*Lycopersicon lycopersicum* (L.) Karst. ex Farw. [syn. *Lycopersicon esculentum* Mill.]	Fruit yield	2.5	9.9	MS
Tomato, cherry	*L. lycopersicum* var. *Cerasiforme* (Dunal) Alef.	Fruit yield	1.7	9.1	MS
Turnip Turnip (greens)	*Brassica rapa* L. (Rapifera Group)	Storage root Top FW	0.9 3.3	9.0 4.3	MS MT
Watermelon	*Citrullus lanatus* (Thunb.) Matsum. & Nakai	Fruit yield	--	--	MS*
Winged bean	*Psophocarpus tetragonolobus* L. DC	Shoot DW	--	--	MT

† These data serve only as a guideline to relative tolerances among crops. Absolute tolerances vary, depending upon climate, soil conditions, and cultural practices.

‡ Botanical and common names follow the convention of Hortus Third (Bailey, 1976) where possible.

§ In gypsiferous soils, plants will tolerate EC_e's about 2 dS/m higher than indicated.

¶ Ratings are defined by the boundaries in Figure 2. Ratings with an * are estimates.

Less tolerant during seedling stage, EC_e at this stage should not exceed 4 or 5 dS/m.

†† Unpublished U. S. Salinity Laboratory data.

‡‡ Grain and forage yields of DeKalb XL-75 grown on an organic muck soil decreased about 26% per dS/m above a threshold of 1.9 dS/m.

§§ Because paddy rice is grown under flooded conditions, values refer to the EC of the soil water while the plants are submerged. Less tolerant during seedling stage.

¶¶ Sesame cultivars, Sesaco 7 and 8, may be more tolerant than indicated by the S rating.

Sensitive during germination and emergence, EC_e should not exceed 3 dS/m.

††† Data from one cultivar, "Probred".

‡‡‡ Average of several varieties. Suwannee and Coastal are about 20% more tolerant, and common and Greenfield are about 20% less tolerant than the average.

§§§ Average for Boer, Wilman, Sand, and Weeping cultivars. Lehmann seems about 50% more tolerant.

A.2 Salt Tolerance of Woody Crops.†

Source: Adapted from Maas and Grattan (1999).

Crop			Salt Tolerance Parameters		
Common Name	**Botanical Name‡**	**Tolerance Based On:**	**Threshold§ (ec_e)**	**Slope**	**Rating¶**
			dS/m	% per dS/m	
Almond	*Prunus duclis* (Mill.) D.A. Webb	Shoot growth	1.5	19	S
Apple	*Malus sylvestris* Mill.		--	--	S
Apricot	*Prunus armeniaca* L.	Shoot growth	1.6	24	S
Avocado	*Persea americana* Mill.	Shoot growth	--	--	S
Banana	*Musa acuminata* Colla	Fruit yield	--	--	S
Blackberry	*Rubus macropetalus* Doug. ex Hook	Fruit yield	1.5	22	S
Boysenberry	*Rubus ursinus* Cham. and Schlechtend	Fruit yield	1.5	22	S
Castorbean	*Ricinus communis* L.		--	--	MS*
Cherimoya	*Annona cherimola* Mill.	Foliar injury	--	--	S
Cherry, sweet	*Prunus avium* L.	Foliar injury	--	--	S*
Cherry, sand	*Prunus besseyi* L., H. Baley	Foliar injury, stem growth	--	--	S*
Coconut	*Cocos nucifera* L.		--	--	MT*
Currant	*Ribes sp.* L.	Foliar injury, stem growth	--	--	S*
Date palm	*Phoenix dactylifera* L.	Fruit yield	4.0	3.6	T
Fig	*Ficus carica* L.	Plant DW	--	--	MT*
Gooseberry	*Ribes sp.* L.		--	--	S*
Grape	*Vitis vinifera* L.	Shoot growth	1.5	9.6	MS
Grapefruit	*Citrus* x *paradisi* Macfady.	Fruit yield	1.2	13.5	S
Guava	*Psidium guajava* L.	Shoot & root growth	4.7	9.8	MT
Guayule	*Parthenium argentatum* A. Gray	Shoot DW Rubber yield	8.7 7.8	11.6 10.8	T T
Jambolan plum	*Syzygium cumini* L.	Shoot growth	--	--	MT
Jojoba	*Simmondsia chinensis* (Link) C. K. Schneid	Shoot growth	--	--	T
Jujube, Indian	*Ziziphus mauritiana* Lam.	Fruit yield	--	--	MT
Lemon	*Citrus limon* (L.) Burm. f.	Fruit yield	1.5	12.8	S
Lime	*Citrus aurantiifolia* (Christm.) Swingle		--	--	S*
Loquat	*Eriobotrya japonica* (Thunb). Lindl.	Foliar injury	--	--	S*
Macadamia	*Macadamia integrifolia* Maiden & Betche	Seedling growth	--	--	MS*
Mandarin orange; tangerine	*Citrus reticulata* Blanco	Shoot growth	--	--	S*
Mango	*Mangifera indica* L.	Foliar injury	--	--	S
Natal plum	*Carissa grandiflora* (E.H. Mey.) A. DC.	Shoot growth	--	--	T

Crop			Salt Tolerance Parameters		
Common Name	**Botanical Name**‡	**Tolerance Based On:**	**Threshold**§ **(ec_e)**	**Slope**	**Rating**¶
Olive	*Olea europaea* L.	Seedling growth, Fruit yield	--	--	MT
Orange	*Citrus sinensis* (L.) Osbeck	Fruit yield	1.3	13.1	S
Papaya	*Carica papaya* L.	Seedling growth, foliar injury	--	--	MS
Passion fruit	*Passiflora edulis* Sims.		--	--	S*
Peach	*Prunus persica* (L.) Batsch	Shoot growth, Fruit yield	1.7	21	S
Pear	*Pyrus communis* L.		--	--	S*
Pecan	*Carya illinoinensis* (Wangenh.) C. Koch	Nut yield, trunk growth	--	--	MS
Persimmon	*Diospyros virginiana* L.		-	--	S*
Pineapple	*Ananas comosus* (L.) Merrill	Shoot DW	--	--	MT
Pistachio	*Pistacia vera* L.	Shoot growth	--	--	MS
Plum; Prune	*Prunus domestica* L.	Fruit yield	2.6	31	MS
Pomegranate	*Punica granatum* L.	Shoot growth	--	--	MS
Popinac, white	*Leucaena leucocephala* (Lam.) de Wit [syn. *Leucaena glauca* Benth.]	Shoot DW	--	--	MS
Pummelo	*Citrus maxima* (Burm.)	Foliar injury	--	--	S*
Raspberry	*Rubus idaeus* L.	Fruit yield	--	--	S
Rose apple	*Syzygium jambos* (L.) Alston	Foliar injury	--	--	S*
Sapote, white	*Casimiroa edulis* Llave	Foliar injury	--	--	S*
Scarlet wisteria	*Sesbania grandiflora*	Shoot DW	--	--	MT
Tamarugo	*Prosopis tamarugo* Phil.	Observation	--	--	T
Walnut	*Juglans* spp.	Foliar injury	--	--	S*

† These data serve only as a guideline to relative tolerances among crops. Absolute tolerances vary, depending upon climate, soil conditions, and cultural practices. The data are applicable when rootstocks are used that do not accumulate Na^+ or Cl^- rapidly or when these ions do not predominate in the soil.

‡ Botanical and common names follow the convention of Hortus Third (Liberty Hyde Bailey Hortorium Staff, 1976) where possible.

§ In gypsiferous soils, plants will tolerate EC_e's about 2 dS/m higher than indicated.

¶ Ratings are defined by the boundaries in Figure 2. Ratings with an * are estimates.

A.3 Boron Tolerance Limits for Agricultural Crops. Threshold Based on Boron Concentration in Soil Water.

Source: Adapted from Grieve et al. (2012).

Crop			Boron Tolerance Parameters		
Common Name	**Botanical Name**	**Tolerance Based On:**	**Threshold† (mg/L)**	**Slope % per mg/L**	**Rating‡**
Alfalfa	*Medicago sativa* L.	Shoot DW	4.0-6.0		T
Apricot	*Prunus armeniaca* L.	Leaf & stem injury	0.5-0.75		S
Artichoke, globe	*Cynara scolymus* L.	Laminae DW	2.0-4.0		MT
Artichoke, Jerusalem	*Helianthus tuberosus* L.	Whole plant DW	0.75-1.0		S
Asparagus	*Asparagus officinalis* L.	Shoot DW	10.0-15.0		VT
Avocado	*Persea americana* Mill.	Foliar injury	0.5-0.75		S
Barley	*Hordeum vulgare* L.	Grain yield	3.4	4.4	MT
Bean, kidney	*Phaseolus vulgaris* L.	Whole plant DW	0.75-1.0		S
Bean, lima	*Phaseolus lunatus* L.	Whole plant DW	0.75-1.0		S
Bean, mung	*Vigna radiata* (L.) R. Wilcz.	Shoot length	0.75-1.0		S
Bean, snap	*Phaseolus vulgaris* L.	Pod yield	1.0	12	S
Beet, red	*Beta vulgaris* L.	Root DW	4.0-6.0		T
Blackberry	*Rubus sp.* L.	Whole plant DW	< 0.5		VS
Bluegrass, Kentucky	*Poa pratensis* L.	Leaf DW	2.0-4.0		MT
Broccoli	*Brassica oleracea* L. (Botrytis group).	Head FW	1.0	1.8	MS
Cabbage	*Brassica oleracea* L. (capitata group)	Whole plant DW	2.0-4.0		MT
Carrot	*Daucus carota* L.	Root DW	1.0-2.0		MS
Cauliflower	*Brassica oleracea* L. (Botrytis group)	Curd FW	4.0	1.9	MT
Celery	*Apium graveolens* L. var. *dulce* (Mill.) Pers.	Petiole FW	9.8	3.2	VT
Cherry	*Prunus avium* L.	Whole plant DW	0.5-0.75		S
Clover, sweet	*Melilotus indica* All.	Whole plant DW	2.0-4.0		MT
Corn	*Zea mays* L.	Shoot DW	2.0-4.0		MT
Cotton	*Gossypium hirsutum* L.	Boll DW	6.0-10.0		VT
Cowpea	*Vigna unguiculata* (L.) Walp.	Seed yield	2.5	12	MT
Cucumber	*Cucumis sativus* L.	Shoot DW	1.0-2.0		MS
Fig, kadota	*Ficus carica* L.	Whole plant DW	0.5-0.75		S
Garlic	*Allium sativum* L.	Bulb yield	4.3	2.7	T
Grape	*Vitis vinifera* L.	Whole plant DW	0.5-0.75		S
Grapefruit	*Citrus x paradisi* Macfady.	Foliar injury	0.5-0.75		S
Lemon	*Citrus limon* (L.) Burm. f.	Foliar injury, Plant DW	< 0.5		VS
Lettuce	*Lactuca sativa* L.	Head FW	1.3	1.7	MS
Lupine	*Lupinus hartwegii* Lindl.	Whole plant DW	0.75-1.0		S

Crop			Boron Tolerance Parameters		
Common Name	**Botanical Name**	**Tolerance Based On:**	**Threshold† (mg/L)**	**Slope % per mg/L**	**Rating‡**
Muskmelon	*Cucumis melo* L. (Reticulatus group)	Shoot DW	2.0-4.0		MT
Mustard	*Brassica juncea* Coss.	Whole plant DW	2.0-4.0		MT
Oats	*Avena sativa* L.	Grain (immature) DW	2.0-4.0		MT
Onion	*Allium cepa* L.	Bulb yield	8.9	1.9	VT
Orange	*Citrus sinensis* (L.) Osbeck	Foliar injury	0.5-0.75		S
Parsley	*Petroselinum crispum* Nym.	Whole plant DW	4.0-6.0		T
Pea	*Pisum sativa* L.	Whole plant DW	1.0-2.0		MS
Peach	*Prunus persica* (L.) Batsch.	Whole plant DW	0.5-0.75		S
Peanut	*Arachis hypogaea* L.	Seed yield	0.75-1.0		S
Pecan	*Carya illinoinensis* (Wangenh.) C. Koch	Foliar injury	0.5-0.75		S
Pepper, red	*Capsicum annuum* L.	Fruit yield	1.0-2.0		MS
Persimmon	*Diospyros kaki* L. f.	Whole plant DW	0.5-0.75		S
Plum	*Prunus domestica* L.	Leaf & stem injury	0.5-0.75		S
Potato	*Solanum tuberosum* L.	Tuber DW	1.0-2.0		MS
Radish	*Raphanus sativus* L.	Root FW	1.0	1.4	MS
Sesame	*Sesamum indicum* L.	Foliar injury	0.75-1.0		S
Sorghum	*Sorghum bicolor* (L.) Moench	Grain yield	7.4	4.7	VT
Squash, scallop	*Cucurbita pepo* L. var *melopepo* (L.) Alef.	Fruit yield	4.9	9.8	T
Squash, winter	*Cucurbita moschata* Poir	Fruit yield	1.0	4.3	MS
Squash, zucchini	*Cucurbita pepo* L. var *melopepo* (L.) Alef.	Fruit yield	2.7	5.2	MT
Strawberry	*Fragaria sp.* L.	Whole plant DW	0.75-1.0		S
Sugar beet	*Beta vulgaris* L.	Storage root FW	4.9	4.1	T
Sunflower	*Helianthus annuus* L.	Seed yield	0.75-1.0		S
Sweet potato	*Ipomoea batatas* (L.) Lam.	Root DW	0.75-1.0		S
Tobacco	*Nicotiana tabacum* L.	Laminae DW	2.0-4.0		MT
Tomato	*Lycopersicon lycopersicum* (L.) Karst. ex Farw.	Fruit yield	5.7	3.4	T
Turnip	*Brassica rapa* L. (Rapifera group)	Root DW	2.0-4.0		MT
Vetch, purple	*Vicia benghalensis* L.	Whole plant DW	4.0-6.0		T
Walnut	*Juglans regia* L.	Foliar injury	0.5-0.75		S
Wheat	*Triticum aestivum* L.	Grain yield	0.75-1.0	3.3	S

† Maximum permissible concentration in soil water without yield reduction. Boron tolerances may vary, depending upon climate, soil conditions, and crop varieties.

‡ The B tolerance ratings are based on the following threshold concentration ranges: < 0.5 mg/L very sensitive (VS), 0.5-1.0 sensitive (S), 1.0-2.0 moderately sensitive (MS), 2.0-4.0 moderately tolerant (MT), 4.0-6.0 tolerant (T), and > 6.0 very tolerant (VT).

A.4 Removal Heavy Metals Techniques

	Technique	Advantages	Drawbacks	Requirements / Comments
Wastewater	Chemical precipitation	-Lower costs -Easy to operate	-Large amount of sludge generation -Filtration or sedimentation processes are necessary - Large quantities of precipitating agents are needed.	-Use of precipitating agent (usually hydroxides or sulfides) -pH of wastewater must be adjusted to the basic conditions at the start
	Chemical coagulation and flocculation	-Easy to operate	-Sedimentation processes are necessary -Need for additional treatments for complete removal. -High operational costs due large amount of chemicals.	-Use of coagulants (alum, ferric chloride, ferrous sulfate, etc.) and flocculants (poly-aluminum chloride, polyacrylamide or polyferric sulfate)
	Electrochemical methods	-Recovery of heavy metals in the elemental metallic state -Additional removal of other compounds (dyes, fluorides, nitrates, sulfates, pharmaceuticals or phenolic compounds) -Lower quantity of sludge generation	-Higher costs (investment and power supply)	-Electrocoagulation, electrodeposition, and electroflotation methods
	Membrane filtration	-Higher removal efficiency -Small operating spaces -Easy to operate	-Higher costs -Membrane fouling -Lower permeate flux	-Exists different types of membranes, including reverse osmosis, ultrafiltration, nanofiltration, and electrodialysis
	Ion exchange	-Higher removal efficiency -Higher treatment capacity -Rapid kinetics	-Higher cost -Large amount of resin is required for high volumes of wastewater	-Usually synthetic resins are used for industrial scale levels. The most used is cation exchange
	Bioremediation	-Lower costs -Easy processes	-Long time is necessary	Usually bioremediation and phytoremediation
	Adsorption	-Higher removal efficiency -Medium-Lower costs -Selective treatment	- Complete saturation of the adsorbent	The most widely adsorbent is activated carbon (AC).

	Technique	Advantages	Drawbacks	Requirements / Comments
		-Regeneration capacity of adsorbent (reversible processes)		
Soil	Replacement, removal or soil isolation	-Higher removal efficiency -High soil quality	-High costs (manpower and material resources) -Only for small areas	-Only when there is no other possible solution.
	Thermal desorption	-Higher removal efficiency	-High cost due high energy consumption	
	Soil leaching or washing	-Medium removal efficiency -Reduced need for additional treatments -Heavy metals can be recycled	-Medium-high cost when specific reagents are used.	-Usually use of fresh water, reagents and other fluids
	Immobilization (Stabilization, vitrification or electrokinetic)	-Lower-medium costs -Avoid the migration of heavy metals to water, plant and other environmental media	-Only temporary solution (contaminants are still in the environment) -Reversible process when soil properties change -Only for soil surface (30-50 cm) -Permanent monitoring is necessary	Electrokinetic is effective in soils with low permeability
	Phytoremediation (Phytostabilization, phytovolatilization or phytoextraction)	-Useful in large contaminated sites -Minimizing the generation of secondary wastes	-More than one growing season is required -Limited to soils less than one meter from the surface and groundwater <3 m from the surface -Type of plants used are limited for climate and hydrologic conditions	-Hyperaccumulators are recommended
	Biological remediation	-Lower costs -Simple treatment	-Long times is required -Difficult to determine whether contaminants are been completely destroyed. -Limited to some microorganisms	

A.5 Concentrations of Selected CECs in Soil, Irrigation Water and Plant Organs of Different Plant Species

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
Pharmaceuticals and Personal Care Products (PPCPs)								
Carbamazepine	Fields	Mixture of Surface Water from River and Groundwater	1.7		Cabbage	Root	61.4	Riemenschneider et al. 2016
						Leaf	79	
						Fruit	9.8	
					Eggplant	Root	192.6	
						Shoot	14	
						Leaf	77.6	
						Fruit	32.2	
					Zucchini	Root	69	
						Shoot	9.3	
						Leaf	41.9	
						Fruit	6.8	
					Tomato	Root	26.7	
						Shoot	40.9	
						Fruit	5	
					Pepper	Root	40	
						Shoot	30.2	
						Fruit	8.3	
					Rucola	Root	37.6	
						Shoot	7.5	
						Leaf	60.7	
					Parsley	Roots	40.8	
						Leaf	90.6	
					Lettuce	Roots	26.7	
						Leaf	215.7	
					Potato	Root	76.6	
						Shoot	59.6	
						Leaf	173.1	
					Carrot	Root	13.9	
						Leaf	61.2	
Carbamazepine	Greenhouse (110 Day)	Wastewater	1.1		Soybean	Root	2.4	Wu et al. 2014
						Stem	0.6	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
						Leaf	1.9	
						Bean	-	
Carbamazepine	Fields	Treated Wastewater (Premature/ Mature)	0.0042		Celery	Stem	0.01 / -	Wu et al.[222]
						Root	0.04 / 0.01	
					Lettuce	Leaf	0.02 / 0.03	
						Root	- / -	
					Cabbage	Leaf	0.04 / -	
						Root	0.05 / 0.02	
						External Leaf	- / 0.04	
					Spinach	Leaf	0.01 / 0.01	
						Root	0.01 / -	
					Carrot	Root	- / -	
					Cucumber	Fruit	- / 0.02	
						Root	- / -	
					Bell Pepper	Fruit	- / -	
						Root	- / 0.01	
					Tomato	Fruit	- / -	
						Root	- / -	
						Stem	- / -	
						Leaf	0.01 / -	
		Fortified Water (Premature/ Mature)	0.225		Celery	Stem	0.64 / 0.4	
						Root	1.8 / 0.6	
					Lettuce	Leaf	2.5 / 1.4	
						Root	1.6 / 1.0	
					Cabbage	Leaf	2.4 / 0.18	
						Root	1.9 / 0.74	
						External Leaf	- / 2.5	
					Spinach	Leaf	0.16 / 0.09	
						Root	1.4 / 0.25	
					Carrot	Root	0.29 / 0.21	
					Cucumber	Fruit	0.46 / 0.51	
						Root	0.44 / 1.6	
					Bell Pepper	Fruit	0.09 / 0.35	
						Root	1.6 / 1.9	
					Tomato	Fruit	- / 0.19	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
						Root	0.95 / 0.50	
						Stem	0.18 / 0.22	
						Leaf	2.1 / 2.7	
Carbamazepine	Greenhouse (70 Day)	Spiked Water	-	0	Lettuce	Root	-	
						Leaf	-	Hurtado et al. 2016
			0.85	4		Root	142	
						Leaf	233	
			10.4	10		Root	234	
						Leaf	461	
			37	20		Root	473	
						Leaf	1031	
			117	40		Root	1214	
						Leaf	2054	
Carbamazepine	Fields (Carrot 100 Days; Potato 154 Days)	Spiked Treated Wastewater	1.35		Sweet Potato*	Root	0.116	Malchi et al. 2014
						Leaf	0.177	
					Carrot*	Root	0.799	
						Leaf	1.069	
Carbamazepine	Field Greenhouse	Ground Water	0.060–0.061		Green Bean	Pod	53.93	Calderón-Preciado et al. 2013
						Leaf	-	
						Root	-	
		Reclaimed Water	0.123–0.369		Carrot		52	
					Green Bean	Pod	114.8	
						Leaf	36.5	
						Root	-	
Carbamazepine	Fields	Surface Water	0.13		Apple Tree	Leaf	0.043	Calderón-Preciado et al. 2011b
					Alfalfa		0.024	
Carbamazepine	Greenhouse-Sandy	Freshwater Spiked with Carbamazepine	8.388	25	Cucumber*		25.6	Shenker et al. 2011
Carbamazepine	Greenhouse-Clay		1.638	25			17.1	
Carbamazepine	Greenhouse-Peat Mixture		0.342	25			6.4	
Carbamazepine	Sandy Soil	Fresh Water: Spiked	0.624	1.15		Xylem Sap (µg/L)	0.33	
						Leaves	18.5	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
						Stems	1.4	
						Roots	3.5	
						Fruits	1.2	
		Reclaimed Wastewater: Not Spiked	0.714	2.99		Xylem Sap (µg/L)	0.52	
						Leaves	20.4	
						Stems	1.1	
						Roots	2	
						Fruits	1	
		Reclaimed Wastewater: Spiked	1.176	4.14		Xylem Sap (µg/L)	1.34	
						Leaves	39.1	
						Stems	1.9	
						Roots	4.5	
						Fruits	2.1	
Caffeine	Fields	Mixture of Surface Water from River and Groundwater	1.3		Cabbage	Roots	32.9	Riemenschneider et al. 2016
						Fruits	21.3	
					Eggplant	Shoots	27.3	
						Leaves	36.8	
					Zucchini	Roots	169	
						Leaves	23.7	
					Tomato	Roots	19.2	
						Shoots	33.4	
					Pepper	Roots	10.3	
						Shoots	13.6	
					Potato	Roots	30.3	
						Shoots	34.2	
						Leaves	61.8	
Caffeine	Fields	Treated Wastewater (Premature/ Mature)	0.011		Celery	Stem	- / 0.17	Wu et al. 2014
						Root	- / 0.25	
					Carrot	Root	- / 0.43	
					Cabbage	Root	0.88 / -	
						External Leaf	- / 0.26	
		Fortified Water	0.219		Celery	Stem	- / 0.8	
						Root	1.2 / -	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
		(Premature/ Mature)			Carrot	Root	- / 1.8	
					Cabbage	Root	1.1 / 0.12	
						External Leaf	- / 1.2	
Caffeine	Greenhouse (70 Day)	Spiked Water	1.5	0	Lettuce	Root	-	Hurtado et al. 2016
						Leaf	-	
			4.2	4		Root	32	
						Leaf	32	
			5.8	10		Root	126	
						Leaf	53	
			18	20		Root	255	
						Leaf	77	
			64	40		Root	398	
						Leaf	147	
Caffeine	Fields (Carrot 100 Days; Potato 154 Days)	Spiked Treated Wastewater	1.55		Sweet Potato*	Root	0.256	Malchi et al. 2014
						Leaf	0.719	
					Carrot*	Root	0.293	
						Leaf	0.603	
Caffeine	Fields	Reclaimed Wastewater Influent	0.789		Apple Tree	Leaf	0.016	Calderón-Preciado et al. 2011b
					Alfafa		<10.6	
		Ter River Influent	0.259		Apple Tree	Leaf	15.5	
					Alfafa		13.9	
Caffeine	Fields	Surface Water	0.54		Apple Tree	Leaf	55.4	Calderón-Preciado et al. 2011b
					Alfalfa		38.4	
Naproxen	Fields	Treated Wastewater (Premature/ Mature)	0.00043		Cabbage	Leaf	0.09 / 0.07	Wu et al. 2014
						Root	- / 0.08	
					Carrot	Root	- / -	
					Cucumber	Fruit	- / -	
						Root	- / 0.18	
					Bell Pepper	Fruit	0.05 / -	
					Tomato	Root	- / 0.05	
						Stem	- / -	
						Leaf	- / -	
		Fortified Water	0.18		Cabbage	Leaf	0.43 / 0.26	
						Root	- / 0.31	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
		(Premature/ Mature)			Carrot	Root	0.31 / -	
					Cucumber	Fruit	0.16 / -	
						Root	0.06 / 0.3	
					Bell Pepper	Fruit	0.39 / 0.39	
					Tomato	Root	- / 0.17	
						Stem	- / 0.14	
						Leaf	- / 0.25	
Naproxen	Fields	Reclaimed Wastewater Influent	0.092		Apple Tree	Leaf	<0.011	Calderón-Preciado et al. 2011a
					Alfafa		<0.011	
		Ter River Influent	0.097		Apple Tree	Leaf	<0.011	
					Alfafa		0.014	
Naproxen	Field Greenhouse	Reclaimed Water	0.368–0.576		Carrot		2.0	Calderón-Preciado et al. 2013
					Green Bean	Pod	44.46	
						Leaf	1.2	
						Root	-	
Naproxen	Fields	Surface Water	0.1		Apple Tree	Leaf	0.043	Calderón-Preciado et al. 2011b
					Alfalfa		0.04	
Diclofenac	Fields	Mixture of Surface Water from River and Groundwater	1.3		Eggplant	Fruits	18	Riemenschneider et al. 2016
Diclofenac	Fields (3 Years)	MWTP I WW	0.15	0.03557	Tomato	Fruit	3.863	Christou et al. 2017
		MWTP II WW	0.09	0.04963			11.615	
Diclofenac	Fields	Surface Water	0.35		Apple Tree	Leaf	0.354	Calderón-Preciado et al. 2011b
					Alfalfa		0.198	
Ciprofloxacin	Fields	Mixture of Surface Water from River and Groundwater	0.3		Cabbage	Fruits	6.7	Riemenschneider et al. 2016
					Carrot	Roots	12	
Diphenhydramine	Greenhouse (110 Day)	Wastewater	0.9		Soybean	Root	1.8	Wu et al. 2014
						Stem	-	
						Leaf	-	
						Bean	-	
Triclosan	Greenhouse (110 Day)	Wastewater	-		Soybean	Root	24.2	Wu et al. 2014
						Stem	58.0	
						Leaf	80.1	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
						Bean	35.8	
Triclosan	Greenhouse (70 Day)	Spiked Water	-	0	Lettuce	Root	-	Hurtado et al. 2016
						Leaf	-	
			0.01	4		Root	21	
						Leaf	13	
			0.056	10		root	147	
						leaf	170	
			0.097	20		Root	353	
						Leaf	25	
			0.167	40		Root	772	
						Leaf	32	
Triclosan	Fields	Surface Water	<0.022		Apple Tree	Leaf	0.043	Calderón-Preciado et al. 2011b
					Alfalfa		0.024	
Triclosan	Greenhouse (110 Day)	Wastewater	2.4		Soybean	Root	7.1	Wu et al. 2014
						Stem	4.8	
						Leaf	14.9	
						Bean	4.0	
Triclosan	Fields	Treated Wastewater (Premature/ Mature)	0.00043		Celery	Root	- / -	Wu et al. 2014
					Lettuce	Root	- / -	
					Cabbage	Root	- / -	
					Spinach	Root	- / -	
					Cucumber	Root	- / -	
					Bell Pepper	Root	- / -	
					Tomato	Root	- / -	
		Fortified Water (Premature/ Mature)	0.18		Celery	Root	2.4 / 1.9	
					Lettuce	Root	1.5 / 3.5	
					Cabbage	Root	3.9 / 2.2	
					Spinach	Root	0.34 / 0.18	
					Cucumber	Root	0.65 / 4.2	
					Bell Pepper	Root	3.4 / 5.0	
					Tomato	Root	0.25 / 0.24	
Sulfamethoxazole	Fields (3 Years)	MWTP I WW	0.64	0.05523	Tomato	Fruit	0.406	Christou et al. 2017
		MWTP II WW	0.98	0.03843			5.255	
Trimethoprim	Fields (3 Years)	MWTP I WW	0.15	0.0467	Tomato	Fruit	0.572	Christou et al. 2017
		MWTP II WW	0.53	0.03243			3.399	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
Ibuprofen	Greenhouse (70 Day)	Spiked Water	-	0	Lettuce	Root	-	Hurtado et al. 2016
						Leaf	-	
			0.73	4		Root	-	
						Leaf	0.93	
			2.1	10		Root	13	
						Leaf	2.4	
			8.7	20		Root	69	
						Leaf	4.9	
			24	40		Root	223	
						Leaf	24	
Ibuprofen	Fields	WWTP Influent	4.299		Apple Tree	Leaf	< 0.012	Calderón-Preciado et al. 2011b
					Alfafa		0.032	
		Ter River Influent	3.54		Apple Tree	Leaf	<0.012	
					Alfafa		0.043	
Ibuprofen	Greenhouse	Ground Water	<LOQ–0.043		Lettuce*		6	Calderón-Preciado et al. 2013
					Green Bean*	Pod	<LOQ	
						Leaf	5.3	
						Root	12	
		Reclaimed Water	0.074–0.35		Lettuce*		5	
					Green* Bean	Pod	2.8	
						Leaf	3.9	
						Root	6.5	
Tetracycline	Field in Huizhou	Wastewater	8.9 (Soil Depth of 0-10cm)		Chinese White Cabbage	Leaf	5.5	Pan et al. 2014
					Rice	Fruit	5.6	
Tetracycline	Field in Foshan		17.1 (Soil Depth of 0-10cm)		Chinese White Cabbage	Leaf	6.3	
						Root	4.2	
					Corn	Fruit	6.6	
						Stem	4.4	
					Rice	Fruit	8.0	
Tetracycline	Field in Zhongshan		15.8 (Soil Depth of 0-10cm)		Chinese White Cabbage	Leaf	4.9	
					Rice	Fruit	8.5	
						Stem	4.8	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (µg/kg)	Conc. in Water (µg/L)	Plant Species	Uptaken Part	Conc. in Plant (µg/kg dw)	Reference
Tetracycline	Field in Guangzhou		13.0 (Soil Depth of 0-10cm)		Chinese White Cabbage	Leaf	4.0	
Tetracycline	Field in Dongguan		21.9 (Soil Depth of 0-10cm)		Chinese White Cabbage	Leaf	10.1	
						Root	5.9	
					Water Spinach	Leaf	6.3	
						Root	4.8	
					Chinese Radish	Leaf	9.2	
						Root	6.5	
Tetracycline	Field in Shenzhen		18.2 (Soil Depth of 0-10cm)		Chinese White Cabbage	Leaf	5.3	
Per- and Polyfluoroalkyl Substances (PFAS)								
PFAS	Greenhouse	Reclaimed Water with Spiked PFASs		0.2	Lettuce	Leaf	204.9	Blaine et al. 2014b
				0.4	Lettuce	Leaf	712	
					Strawberry	Root	337.22	
						Shoot	322.79	
				1	Lettuce	Leaf	1564.1	
				2	Lettuce	Leaf	2348	
				4	Lettuce	Leaf	5966	
					Strawberry	Root	2841	
						Shoot	703.78	
				10	Lettuce	Leaf	12343	
				20	Lettuce	Leaf	28750	
				40	Lettuce	Leaf	58970	
					Strawberry	Root	19404	
						Shoot	10711.2	
PFAS	Greenhouse	Groundwater	62 (Soil Depth of 5 cm)	37.6	Tomato		105	Bao et al. 2020
					Cucumber		82	
Plasticiser								
Bisphenol A	Greenhouse (70 Day)	Spiked Water	-	0	Lettuce	Root	-	Hurtado et al. 2016
						Leaf	-	
			0.0051	4		Root	73	
						Leaf	33	

Chemical Type	Growth Conditions	Irrigation Source	Conc. in Soil (μg/kg)	Conc. in Water (μg/L)	Plant Species	Uptaken Part	Conc. in Plant (μg/kg dw)	Reference
			0.011	10		Root	124	
						Leaf	54	
			0.025	20		Root	212	
						Leaf	83	
			0.055	40		Root	325	
						Leaf	158	
Flame Retardant								
Tributyl Phosphate	Field Greenhouse	Ground Water	<LOQ		Lettuce*		31	Calderón-Preciado et al. 2013
					Carrot*		10	
		Reclaimed Water	<LOQ		Lettuce*		188	
					Carrot*		10	
					Green* Bean	Pod	1.82	
						Leaf	6.3	
						Root	6.7	

* Concentration based on fresh weight

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