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61

by
Kenneth K. Tanji
Hydrology Program
Department of Land, Air and Water Resources
University of California
Davis, United States of America

Neeltje C. Kielen
Water Resources, Development and Management Service
FAO Land and Water Development Division

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Foreword

Irrigated agriculture has made a significant contribution towards world food security. However, water resources for agriculture are often overused and misused. The result has been large-scale waterlogging and salinity. In addition, downstream users have found themselves deprived of sufficient water, and there has been much pollution of freshwater resources with contaminated irrigation return flows and deep percolation losses. Irrigated agriculture needs to expand in order to produce sufficient food for the world's growing population. The productivity of water use in agriculture needs to increase in order both to avoid exacerbating the water crisis and to prevent considerable food shortages. As irrigated agriculture requires drainage, a major challenge is to manage agricultural drainage water in a sustainable manner.

Up until about 20 years ago, there were few or indeed no constraints on the disposal of drainage water from irrigated lands. One of the principle reasons for increased constraints on drainage disposal is to protect the quality of receiving waters for downstream uses and to protect the regional environment and ecology. Many developed and developing countries practise drainage water management. This study has brought together case studies on agricultural drainage water management from the United States of America, Central Asia, Egypt, India and Pakistan in order to learn from their experiences and to enable the formulation of guidelines on drainage water management. From the case studies, it was possible to distinguish four broad groups of drainage water management options: water conservation, drainage water reuse, drainage water disposal and drainage water treatment. Each of these options has certain potential impacts on the hydrology and water quality in an area. Interactions and trade-offs occur when more than one option is applied.

Planners, decision-makers and engineers need a framework in order to help them to select from among the various options and to evaluate their impact and contribution towards development goals. Moreover, technical expertise and guidelines on each of the options are required to enable improved assessment of the impact of the different options and to facilitate the preparation of drainage water management plans and designs. The intention of this publication is to provide guidelines to sustain irrigated agriculture and at the same time to protect water resources from the negative impacts of agricultural drainage water disposal.

This publication consists of two parts. Part I deals with the underlying concepts relating to drainage water management. It discusses the adequate identification and definition of the problem for the selection and application of a combination of management options. It then presents technical considerations and details on the four groups of drainage management options. Part II contains the summaries of the case studies from the United States of America, Central Asia, Egypt, India and Pakistan. These case studies represent a cross-section of approaches to agricultural drainage water management. The factors affecting drainage water management include geomorphology, hydrology, climate conditions and the socio-economic and institutional environment. The full texts of the case studies can be found on the attached CD-ROM.

Acknowledgements

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- PC with Intel Pentium® processor and Microsoft® Windows 95 / 98 / 2000 / Me / NT / XP
  or
- Apple Macintosh with PowerPC® processor and Mac OS® 8.6 / 9.0.4 / 9.1 / X
- 64 MB of RAM
- 24 MB of available hard-disk space
- Internet browser such as Netscape® Navigator or Microsoft® Internet Explorer
- Adobe Acrobat® Reader (included on CD-ROM)

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List of acronyms and symbols

Acronym/ symbol	Description	Dimension
ρ	bulk density (kg m^{-3}, g cm^{-3})	M L^{-3}
μ	drainable pore space (m^3 m^{-3})	-
α	empirical shape parameter (cm^{-1})	L^{-1}
λ	empirical shape parameter depending on dK/dh	-
θ	volumetric water content (m^3 m^{-3})	-
θ_{fc}	volumetric water content at field capacity (m^3 m^{-3})	-
Δh	drop in groundwater table (m)	L
ΔM_{ss}	changes in storage of soluble soil salts (kg, t)	M
ΔM_{xc}	changes in storage of exchangeable cations (kg, t)	M
ΔS	change in salt storage in the rootzone (ECmm)	$T^3I^2M^{-1}L^{-2}$
ΔW	change in moisture content in the rootzone (mm)	L
ΔW_{rz}	change in water storage in the rootzone (mm)	L
ΔW_{sz}	change in water storage in the saturated zone (mm)	L
ΔW_{vsz}	change in water storage in the vadose and saturated zone (mm)	L
ΔW_{vz}	change in water storage in the vadose zone (mm)	L
A	salinity threshold (bars)	$ML^{-1}T^{-2}$
a	salinity threshold (dS/m)	$T^3I^2M^{-1}L^{-3}$
AW	applied water (mm)	L
B	slope expressed in percent per bar	-
b	slope expressed in percent per dS/m	$T^3I^2M^{-1}L^{-3}$
B_I	boron concentration in irrigation water (mg $litre^{-1}$)	M L^{-3}
BOD	biochemical oxygen demand (mg $litre^{-1}$)	M L^{-3}
B_{ss}	boron concentration in the soil solution (mg $litre^{-1}$)	M L^{-3}
B_{ss0}	initial boron concentration in the soil solution (mg $litre^{-1}$)	M L^{-3}
B_{sst}	desired boron concentration in the soil solution (mg $litre^{-1}$)	M L^{-3}
C_{dw}	salt concentration of drainage water (kg m^{-3} or mg $litre^{-1}$, g $litre^{-1}$)	M L^{-3}
CEC	cation exchange capacity (meq $100g^{-1}$ or mMol (c) $100g^{-1}$)	N M^{-1}
C_{gw}	salt concentration of groundwater (mg $litre^{-1}$ or kg m^{-3})	M L^{-3}
C_I	salt concentration of irrigation water (kg m^{-3} or mg $litre^{-1}$, g $litre^{-1}$)	M L^{-3}
CIMIS	California Irrigation Management Information System	-
C_{IW}	salt concentration of the infiltrated water (mg/litre)	M L^{-3}
C_k	solute species (mMol $litre^{-1}$)	N L^{-3}
COD	chemical oxygen demand (mg $litre^{-1}$)	M L^{-3}
C_{R^*}	salt concentration of the net percolation water (mg/litre)	M L^{-3}
CVRWQCB	Central Valley Regional Water Quality Control Board	-
$\overline{C}_k$	exchangeable form of solute species(mMol (c) $100g^{-1}$)	N M^{-1}
$\widehat{C}_k$	mineral form of solute species (mMol (c) $100g^{-1}$)	N M^{-1}
D	dispersion coefficient (m^2 d^{-1})	L^2 T^{-1}
D_{rz}	depth of the rootzone (m)	L
D_L	depth of leaching water (mm)	L
D_r	drainage water reuse (mm)	L
D_{ra}	artificial subsurface drainage (mm)	L
D_{rn}	natural drainage (mm)	L
D_s	depth of soil to be reclaimed (mm)	L
E	evaporation (mm)	L
e_a	irrigation application efficiency	-
EC	electrical conductivity (dS m^{-1}, mS cm^{-1})	$T^3I^2M^{-1}L^{-3}$
e_c	water conveyance efficiency	-
EC_0	EC_e at which the crop yield is reduced to zero (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{50}	EC_e at which the crop yield is reduced to 50 percent (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{Dra}	EC of subsurface drainage water (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$

Acronym/ symbol	Description	Dimension
EC_e	EC of soil water of the saturated paste (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{e0}	initial EC_e (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{et}	desired EC_e (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{fc}	EC of soil water at field capacity (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{frR}	EC of percolation water mixed with soil solution (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{gw}	EC of groundwater (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_I	EC of the irrigation water (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{IW}	EC of the infiltrated water (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{IWi}	EC of the infiltrated water mixing with soil solution (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_R	EC of the percolation water (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{Si}	EC of S_i intercepted by subsurface drains (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{SW}	EC of the soil water (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
EC_{ts}	threshold EC of the extract from saturated soil paste (dS m^{-1})	$T^3I^2M^{-1}L^{-3}$
e_d	distribution efficiency	-
ESP	exchangeable sodium percentage	-
ET	evapotranspiration (mm)	L
ET_{crop}	crop evapotranspiration (mm)	L
ETo	reference crop evapotranspiration (mm)	L
f	leaching efficiency coefficient	-
f_i	leaching efficiency coefficient of water mixing with soil solution	-
f_r	leaching efficiency coefficient of the percolation water	-
G	capillary rise (mm)	L
H	hydraulic head (m)	L
h	soil pressure head (m)	L
I	total applied irrigation water (mm)	L
$ICUC$	irrigation consumptive use coefficient	-
IE	irrigation efficiency	-
IFDM	integrated farm drainage management	-
I_g	groundwater irrigation (mm)	L
I_i	infiltrated irrigation water (mm)	L
IS	irrigation sagacity	-
I_s	surface irrigation (mm)	L
IW	infiltrated water (mm)	L
K	hydraulic conductivity (m d^{-1})	$L\,T^{-1}$
K_c	crop coefficient	-
$K(\theta)$	unsaturated hydraulic conductivity (m d^{-1})	$L\,T^{-1}$
K_s	saturated hydraulic conductivity (m d^{-1})	$L\,T^{-1}$
LF	leaching fraction	-
LF_i	leaching fraction of water mixing with the soil solution	-
LR	leaching requirement	-
LR_i	leaching requirement of water mixing with soil solution	-
M_c	mass of salts removed by harvested crops (kg)	M
M_d	mass of salts dissolved from mineral weathering (kg)	M
M_f	mass of salts from fertilisers and amendments (kg)	M
M_p	mass of salts precipitated in soils (kg)	M
MPN	most probable number of faecal coliform	-
M_{Se}	mass of selenium (kg)	M
n	dimensionless empirical shape parameter	-
OP_{fc}	osmotic potential at field capacity (bar)	$ML^{-1}T^{-2}$
O&M	operation and maintenance	-
P	precipitation (mm)	L
P_e	effective precipitation (mm)	L
q	soil water flux or specific discharge (m d^{-1})	$L\,T^{-1}$
R	deep percolation (mm)	L
R^*	net deep percolation (mm)	L
RO	surface runoff (mm)	L
S	salts in rootzone (ECmm)	$T^3I^2M^{-1}L^{-2}$
SAR	sodium adsorption ratio (meq^{½} litre$^{-½}$)	$N^{½}L^{-1½}$
S_c	seepage from canal (mm)	L
SCARP	Salinity Control and Reclamation Project	-

Acronym/ symbol	Description	Dimension
S_e	water extraction sink (m^3 m^{-3} d^{-1})	T^{-1}
Se_{hp}	maximum selenium concentration in harvested product (mg kg^{-1})	-
Se_I	selenium concentration in irrigation water (μg $litre^{-1}$)	$M\ L^{-3}$
S_{end}	salts in the rootzone at the end of the period (ECmm)	$T^3I^2M^{-1}L^{-2}$
S_p	lateral seepage (mm)	L
Se_{ss}	selenium concentration in soil solution (μg $litre^{-1}$)	$M\ L^{-3}$
Se_{ssm}	maximum selenium concentration in soil solution (μg $litre^{-1}$)	$M\ L^{-3}$
S_i	seepage inflow (mm)	L
S_{IW}	salts in infiltrated water (ECmm)	$T^3I^2M^{-1}L^{-2}$
S_R	salts in percolation water from the rootzone (ECmm)	$T^3I^2M^{-1}L^{-2}$
S_{start}	salts in the rootzone at the start of the period (ECmm)	$T^3I^2M^{-1}L^{-2}$
S_v	vertical seepage (mm)	L
t	time (d)	T
TDS	total dissolved solids (mg $litre^{-1}$, g $litre^{-1}$)	$M\ L^{-3}$
TSS	total suspended solids (mg $litre^{-1}$, g $litre^{-1}$)	$M\ L^{-3}$
V_{dw}	volume of drainage water (m^3)	L^3
V_{gw}	volume of groundwater (m^3)	L^3
V_i	volume of irrigation water (m^3)	L^3
Y	empirical correction factor	-
Y_r	relative yield	-
W_{fc}	moisture content at field capacity (mm)	L
WT	water table depth (m)	L
z	vertical coordinate (m)	L

Part I

Framework and technical guidelines

Chapter 1

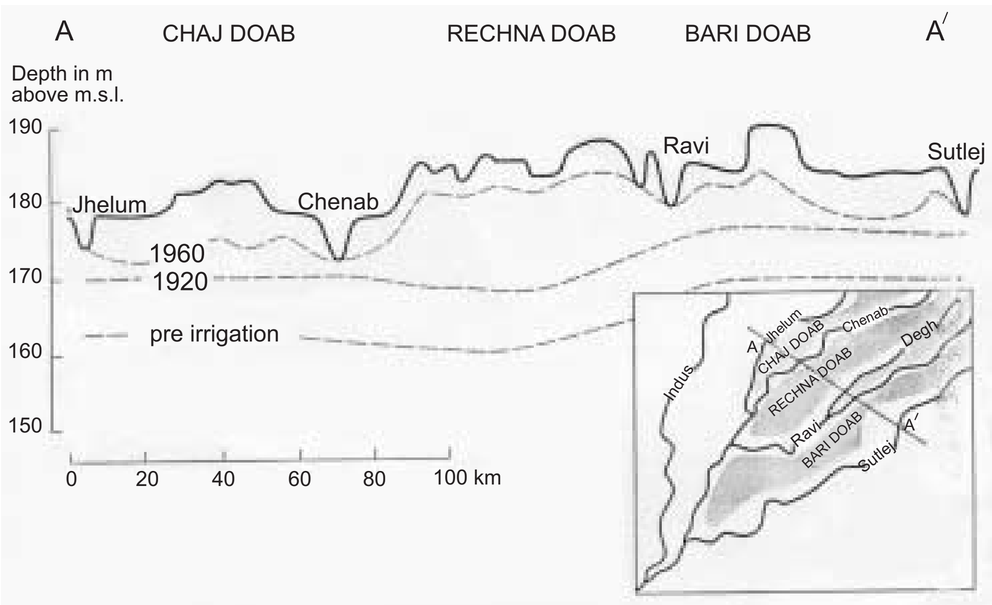
Introduction

NEED FOR DRAINAGE OF IRRIGATED LANDS

Large-scale development of irrigation has taken place in many arid and semi-arid areas since the late nineteenth century. Although irrigation has greatly increased the agricultural production potential, recharge brought about by seepage losses from the irrigation network and deep percolation from farm irrigation has accumulated into the underlying groundwater. A rise in water table results when irrigation-induced recharge is greater than the natural discharge. In many irrigated areas around the world, rising water tables have subsequently led to waterlogging and associated salinity problems. This has happened where drainage development has not kept pace with irrigation development or where maintenance of drainage facilities has largely been neglected. As an example of the historical rise in the groundwater table after the introduction of large-scale irrigation, Figure 1 shows the elevations of the ground surface and the variations of the phreatic level in an irrigated area in Punjab, Pakistan.

Salinization affects about 20–30 million ha of the world's 260 million ha of irrigated land FAO (2000). To maintain favourable moisture conditions for optimal crop growth and to control soil salinity, drainage development is indispensable especially in saline groundwater zones. Smedema *et al*. (2000) estimate that current drainage improvement programmes cover less

FIGURE 1
Rise of the groundwater table in Punjab, Pakistan



Source: Bhutta and Wolters, 2000.

TABLE 1
Salinized and drained areas compared with total irrigated area, Central Asia and the Near East

Country	Irrigated area	Salinized area		Total drained area surface + subsurface drained		% subsurface drainage of irrigated area
	ha	ha	% of irrigated area	ha	% of irrigated area	
Central Asia						
Kazakhstan	3 556 400	242 000	6.8	433 100	12.1	0.4
Kyrgyzstan	1 077 100	60 000	5.6	149 000	13.8	6.1
Tajikistan	719 200	115 000	16.0	328 600	45.7	19.1
Turkmenistan	1 744 100	652 290	37.4	1 022 126	58.6	18.5
Uzbekistan	4 280 600	2 140 550	50.0	2 840 000	66.3	16.3
Near East						
Bahrain	3 165	1 065	33.6	1 300	41.1	
Egypt	3 246 000	1 210 000	37.3	2 931 000	90.3	38.5
Iran	7 264 194	2 100 000	28.9	40 000	0.6	0.6
Jordan	64 300	2 277	3.5	4 000	6.2	
Kuwait	4 770	4 080	85.5	2	0	0
Lebanon	87 500			10 800	12.3	0
Mauritania	49 200			12 784	26.0	
Pakistan	15 729 448			5 100 165	32.4	
Saudi Arabia	1 608 000			44 000	2.7	
Syria	1 013 273	60 000	5.9	273 030	26.9	
Tunisia	385 000			162 000	42.1	42.1
Turkey	4 185 910			3 143 000	75.1	

Source: FAO, 1997a, 1997b.

than 0.5 million ha per year, insufficient in their view to balance the current growth of affected drainage areas. They estimate that: 10–20 percent of the irrigated land is already equipped with drainage; 20-40 percent of the irrigated area is not in need of any artificial drainage; while 40–60 percent is in need of drainage but remains without drainage facilities. Table 1 shows examples from countries in Central Asia and the Near East to illustrate their observations.

In Central Asia, the present drainage infrastructure is insufficient to control irrigation-induced waterlogging and salinity with a comparatively small percentage of subsurface drained land. In addition, the poor state of drainage networks (due to lack of maintenance) has exacerbated waterlogging and salinity (FAO, 1997a).

In the Near East, which is a region subject to salinity problems due to the prevailing climate conditions, an average of about 29 percent of the irrigated areas have salinity problems. Table 1 shows that for 12 countries in the Near East on average about 34 percent of the irrigated area has been provided with drainage facilities. For most countries no figures are available on the area under surface versus subsurface drainage (FAO, 1997b).

In Pakistan, 13 percent of the irrigated area is reportedly suffering from severe salinity problems in spite of the efforts made to provide drainage in irrigated areas. Salinity problems persist because of deficiencies in water policies and the low priority attached to the allocation of resources for the operation and maintenance (O&M) of drainage facilities in favour of initiating new projects (Martínez Beltrán and Kielen, 2000).

On the other hand, under the influence of the growing world population and the increasing demand for food, there is a trend of irrigation intensification. To supplement scarce surface water resources, groundwater is exploited through tubewell development, mainly in fresh groundwater zones, all over the world. In many of these areas, the water table is declining due to overexploitation of groundwater resources. Therefore, problems of waterlogging and related salinization in irrigated agriculture are confined principally to saline groundwater zones. However, the salinization and sodification of agricultural lands resulting from irrigation with marginal and poor-quality water (mainly groundwater) is increasing rapidly. Although firm figures are not currently available, many cases have been reported and documented for major irrigated areas in the world including South Asia, Southeast Asia, Central Asia, North Africa, the Near East, Australia, and the United States of America.

NEED FOR WATER CONSERVATION AND REUSE

In response to the increasing world population and economic growth, water withdrawals for human consumption will increase, so increasing the competition for water between municipal, industrial, agricultural, environmental and recreational needs. If present trends continue with water withdrawal under present practices and policies, it is estimated that by 2025 water stress will increase in more than 60 percent of the world (Cosgrove and Rijsberman, 2000).

In this respect, providing food for the growing population is a major challenge as agriculture is already by far the largest water consumer in most regions in the world, except North America and Europe. On a global basis, agriculture accounts for 69 percent of all water withdrawals (FAO, 2000). Although the water resources for agriculture are often overused and misused, the general belief is that irrigated agriculture has to expand by 20-30 percent in area by 2025 in order to produce sufficient food for the growing world population. In order to avoid exacerbating the water crisis and to prevent considerable food shortages, the productivity of water use needs to increase. In other words, the amount of food produced with the same amount of water needs to increase. This is possible through the conservation and reuse of the available water resources in the agriculture sector, including usable drainage waters.

The overuse and misuse of water in irrigated agriculture has not only resulted in large-scale waterlogging and salinity and overexploitation of groundwater resources, but also in the depriving of downstream users of sufficient water and in the pollution of fresh water resources with contaminated irrigation return flow and deep percolation losses. Water pollution adds to the competition for scarce water resources as it makes them less suitable for other potential beneficial downstream uses. Furthermore, it might cause severe environmental pollution and threaten public health (Box 1).

BOX1: NEED FOR CONSERVATION OF WATER QUALITY – EXAMPLE FROM THE ARAL SEA BASIN

In the Aral Sea Basin, about 37 km^3 of irrigation return water is generated each year. Most of it returns to the river system (16–18 km^3). In most regions, river water also serves domestic, industrial and environmental purposes. Due to the river disposal, the downstream quality of the river water deteriorates. The salt content of the river increases from about 0.5 g/litre in the upstream regions to 1–1.5 g/litre in the delta areas, where saline and polluted drinking-water poses severe health problems to communities. Moreover, in downstream areas the high salt content of the irrigation water, caused by upstream disposals, aggravates the salinity status of the irrigated lands (case study on the Aral Sea Basin in Part II).

TOWARDS DRAINAGE WATER MANAGEMENT

Until ten years ago drainage water management received little attention. Drainage research tended to focus on design issues, while evaluation dealt largely with the performance of the installed system in relation to the design criteria (Snellen, 1997). After the 1992 Earth Summit, the international irrigation and drainage community focused its full attention on drainage water management. Agenda 21 not only stresses the need for drainage as a necessary complement to irrigation development in arid and semi-arid areas, but at the same time it urges the conservation and recycling of freshwater resources in a context of integrated resource management (UNCED, 1992). In many countries these concerns and especially the concern for water quality degradation have resulted in drainage water disposal regulations to maintain the water quality standards of freshwater bodies for other uses, i.e. agricultural, municipal, industrial, environmental and recreational uses.

SCOPE OF THIS PUBLICATION

This publication focuses on the management of drainage water from existing drainage systems located in irrigated areas in arid and semi-arid regions. It does not address design considerations for new drainage systems in detail as these will be the subject of the forthcoming FAO Irrigation and Drainage Paper *Planning and design of land drainage systems*.

For existing drainage facilities, planners, decision-makers and engineers have a number of drainage water management options available to attain their development goals, e.g. reducing the waterlogged and salinized area in a certain drainage basin whilst maintaining water quality for downstream users. The options can be divided into four broad groups of measures: (i) water conservation, (ii) drainage water reuse, (iii) drainage water disposal, and (iv) drainage water treatment. Each of these options has certain potential impacts on the hydrology and water quality in an area, and where more than one option is applied at a site, interactions and trade-offs occur. Therefore, planners, decision-makers and engineers need to have a framework for selecting from among the various possibilities and for evaluating the impact and the contribution towards the development goals. Furthermore, technical expertise and guidelines on each of the options are required to enable enhanced assessment of the impact of the differing options.

The objective of this publication is twofold:

1. To present a framework that will enable planners, decision-makers and engineers to select from among the differing drainage water management options and to evaluate their impact and contribution towards the development goals; and
2. To provide technical guidelines for the planning and preparation of preliminary designs of drainage water management options.

This publication consists of two parts. Part I provides the framework and technical guidelines on the drainage water management measures for decision-makers, planners and engineers. After the introduction (Chapter 1), Chapter 2 presents guidelines for defining the problem and alternative approaches in seeking solutions. Chapter 3 provides a framework for the selection and evaluation of drainage water management options. Chapter 4 deals with factors affecting drainage water quality. Chapters 5 to 8, respectively, present the guidelines and details on each of the four options related to water conservation, drainage water reuse, drainage water disposal and drainage water treatment. It is beyond the scope of this publication to provide technical details and guidelines to prepare detailed designs. Design engineers may refer to the numerous references provided later in the text.

Part II presents summaries of five case studies from India, Pakistan, Egypt, the United States of America, and the Aral Sea Basin. They illustrate how various countries or states deal with the issue of drainage water management in the context of water scarcity, both in quantitative and qualitative terms, under differing degrees of administrative and policy guidelines and regulations as well as differing degrees of technological advancements and possibilities. The attached CD-ROM contains the full text of these case studies.

Chapter 2
Defining the problem and seeking solutions

SYSTEM APPROACH IN DRAINAGE WATER MANAGEMENT

When planners, decision-makers or engineers face the need for a change in drainage water management, the nature of the exact problem is often not clear and the perceived problem depends considerably on the individual's viewpoint. For example, a farmer's perception of a problem related to drainage water management depends on the physical conditions within the farm boundaries and may differ substantially from those of an irrigation district, national water resource authority or environmental pressure group. In such situations, a soft system approach can help define the problem and seek solutions. A key feature of the soft system approach is that it attempts to avoid identifying problems and seeking solutions from only one perspective and excluding others.

The characteristic of a system is that, although it can be divided into subsystems, it functions as a whole to achieve its objectives. The successful functioning of a system depends on how well it satisfies changing external and internal demands. The system itself is part of a broader universe. In all natural or agricultural systems, there exists a hierarchy of levels (Stephens and Hess, 1999). For irrigated crop production this hierarchy might be:

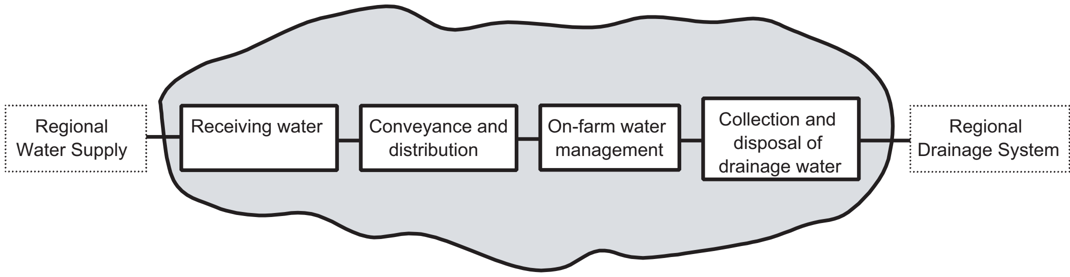
- biochemical and physical systems;
- plant and cropping system;
- farming system;
- irrigation and drainage systems;
- regional, river- or drainage-basin system; and
- supra-regional systems.

Figure 2 provides a simplified example of a water management system within an irrigation district. It shows some main characteristics of a system as described by the Open University (1997).

1. Defining the boundaries of a system is not a simple task. For example, there could be questions as to whether it would be better to include the regional drainage system within the system boundaries and include pressure from environmental groups, other water users and water quality rules and regulations in the system's environment. The boundaries chosen reflect the perception of the systems analysts and their understanding of the system's behaviour, which may not always coincide with an organizational or departmental unit. System boundaries in drainage water management might more often coincide with hydrological units.

2. The elements included in the system's environment are those which influence the system in an important way but over which the system itself has no control because they are driven by forces external to the system of interest.

FIGURE 2
Example of a water management system within an irrigation district



3. The elements within the system are functional working parts of the system. The way that the system operates and behaves depends on the interactions between the elements in the system. Elements could be decomposed into smaller subsystems. The level of detail depends on the specific objectives of the systems analysis. For example, in systems analysis for drainage water management, it would be important to show individual farms as subsystems because on-farm water management defines to a great extent the total quantity and quality of the generated drainage water.

4. As the creator of the system image, a systems analyst defines a particular perspective while another analyst of differing disciplinary training may produce a different image.

When applying a systems approach, at first it is necessary to include all elements in the picture whether they relate to physical, technical, economic, legal, political or administrative considerations as well as any subjective considerations based on the understanding, norms, values and beliefs of the stakeholders involved. A premature exclusion of important elements may result in a suboptimal course of action (Open University, 1997). Figure 3 shows an example of a simplified holistic picture of the on-farm water management subsystem. This subsystem contains a mixture of physical, economical, social, legal and subjective considerations that all have some important influence on on-farm water management.

Where changes in drainage water management are necessary, a simple method to produce predetermined results might not work. Rather, a more open-ended investigative approach is

FIGURE 3
Example of a simplified holistic picture of the on-farm water management sub-system

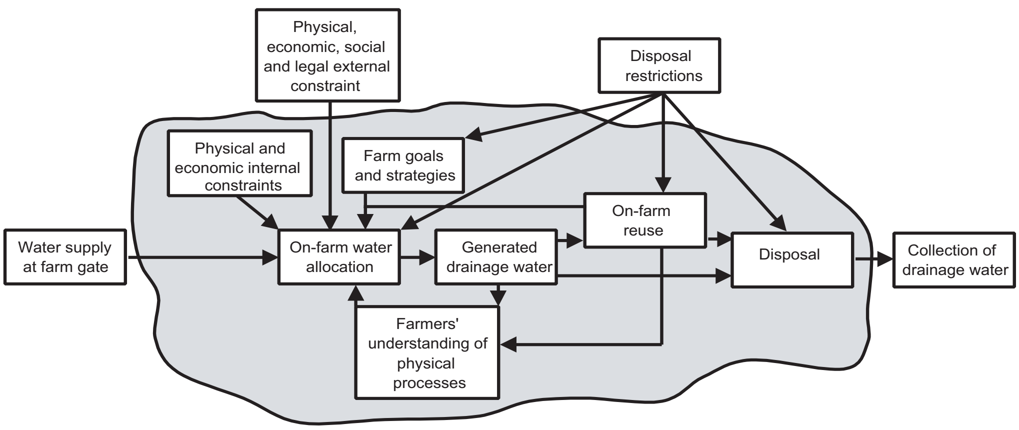
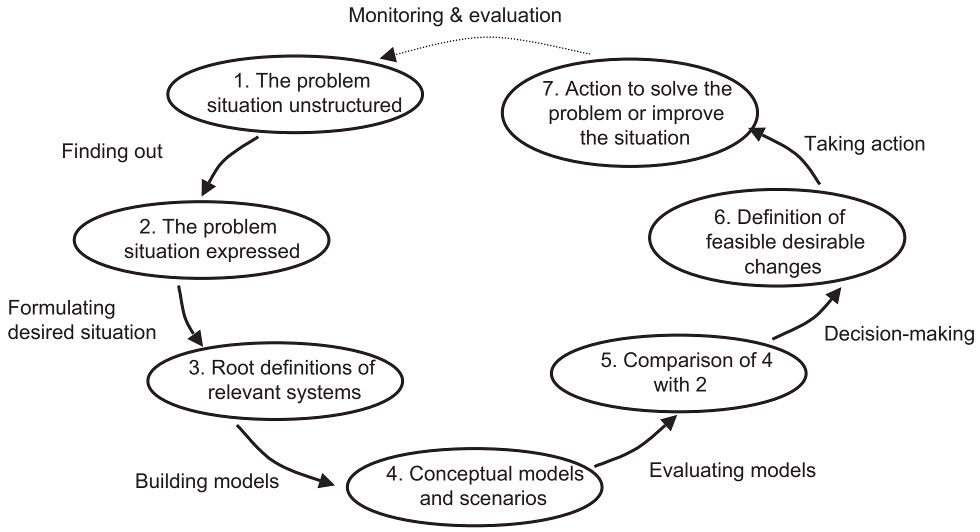


FIGURE 4
The seven stages in the soft system methodology



Source: after Bustard *et al*., 2000.

required in which important avenues are explored and considered and in which there is room for iterative processes. The outcome of such an investigative approach is not necessarily the optimal final solution but rather a solution that seems best for those involved, for that particular time and under those particular circumstances. As the external environment and internal demands change constantly, the mix of solutions and needs for change are changing continuously. Moreover, the achieved changes themselves give new insights into processes and enable a continuous learning process.

The soft system approach to identifying causes and seeking solutions takes the aforementioned considerations into account. The soft system methodology is a seven-stage approach (Checkland, 1981) that has been adopted and adjusted for numerous purposes. Figure 4 presents the steps in the soft system method. The soft system methodology was developed for managing changes in the context of human activity systems, i.e. a set of activities undertaken by people linked together in a logical structure to constitute a purposeful whole (Checkland, 1989). As drainage water management is a complex mix of human activity and natural and designed systems, the soft system approach presented in this chapter is not a direct interpretation of the soft systems methodology introduced by Checkland. Rather it is a free interpretation to illustrate the steps that in general need to be taken to identify the actual problems and search for possible courses of action. The following sections explain the different steps in the context of drainage water management in more detail.

DEFINING THE PROBLEM

Defining the problem and diagnosing the causes is the key to seeking solutions. In the context of drainage water management a problem might be defined as: a need, request or desire for a change in present situation subject to a number of conditions or criteria that must be satisfied simultaneously. This definition implies that a problem situation does not exist independent of the stakeholders who perceive the problem. The solutions to the problem are subject to criteria and conditions based on people's perception of the ideal situation. The conditions and criteria are subjective insofar as they are based on personal or communal objectives influenced by local to national economic, social, cultural, ecological and legal motives, norms and values.

> In the field of agricultural drainage water management, the root cause of the problem(s) nearly always stems from human interference in the natural environment.

Step 1. An analysis of the problem and the search for solutions starts in a situation where someone or a group of people perceives that there is a problem. At this stage, it might not be possible to define the problem with precision, as different people involved will have differing perceptions of the problem.

Step 2. The first step in formulating the problem more precisely is to identify all the stakeholders involved and their relation to the problem. To form what is called the rich picture, all the elements must be included whether they relate to physical, technical, economic, legal, political or administrative considerations along with subjective considerations based on understanding, norms, values and beliefs of the stakeholders involved. It is then necessary to extract areas of conflict or disagreement as well as the key tasks that must be undertaken within the problem situation.

Step 3. Once the problem situation has been analysed and expressed from the key issues, the relevant systems and subsystems can be defined. These systems can be formal or informal and are those that carry out purposeful activities that will lead to improvement or elimination of the problem situation. Examples of a relevant system within the context of drainage water management might be an on-farm water management system, a system for development of water quality rules and regulation for drainage water disposal, or a land-use planning system. An analysis of the various elements involved provides valuable insight into different perspectives on and constraints surrounding the situation. For each relevant system, a root definition can be formulated. A root definition is a formulation of the relevant system and the purpose of the system to achieve a situation in which the problem is balanced out or eliminated. Each root definition provides a particular perspective of the system under investigation. In general, a root definition should include the following information: what the purposeful activity carried out by the system is; who the 'owner' of the system is; who the beneficiaries/victims of the purposeful activity are; who will implement the activity; and what the constraints in its environment are that surround the system (Checkland, 1989). A root definition for on-farm water management might be:

> In an on-farm water management system the responsible farmer uses irrigation water in such a manner that the drainage water generated is of such quantity and quality as permitted by the drainage disposal act whilst maintaining long-term favourable soil conditions that guarantee the production of valuable crops to ensure the financial sustainability of the farming enterprise.

A root definition for the development of water quality rules and regulation could be:

> A governmental system in which water quality rules and regulations for the disposal of agricultural drainage water are promulgated such that they will guarantee water quality for beneficial downstream water uses, including the maintenance of valuable ecosystems, while ensuring the economic sustainability of the agriculture sector.

SEEKING SOLUTIONS

Step 4. On the basis of the root definitions, conceptual models need to be constructed. These models include all the probable activities and measures that the system needs to implement to achieve the root definition. In other words, alternative scenarios need to be formulated. This involves doing sufficient work on the technical and other details, which need to be defined, in order to enable sound decision-making. Moreover, measurable indicators need to be established to compare the results of the conceptual models with the analysed situation or base case. As the formulation of scenarios is based on a thorough analysis of the systems, it should take into consideration system objectives, possibilities and constraints.

> Improvements needed to minimize or correct a particular drainage water management problem may consist of physical structures, non-physical improvements or both. Physical improvements could involve using irrigation water conservatively by on-farm water management practices along with regional drainage practices such as recirculating usable drainage water to meet waste discharge requirements. Non-physical improvements may include implementing tiered water pricing to encourage growers to use water wisely, i.e. charging a penalty for overuse.

Step 5. The next step is to compare the scenarios or conceptual models with the situation analysis. The idea is to test the scenarios and decide whether the implementation of a scenario would resolve the defined key issues.

Step 6. If it would, it needs to be investigated and there needs to be debate as to whether the changes proposed, resulting from implementation of the scenario, are both desirable and feasible. What is desirable and what is feasible might clash as a result of system objectives, possibilities and constraints.

Step 7. The final step is to define the measures and changes to be implemented.

SPATIAL ISSUES

Drainage water problems vary in space and time due mainly to soil heterogeneity and water management practices. The following example illustrates how spatial variability needs to be taken into account in the problem analysis, and also how it influences the options for drainage water management.

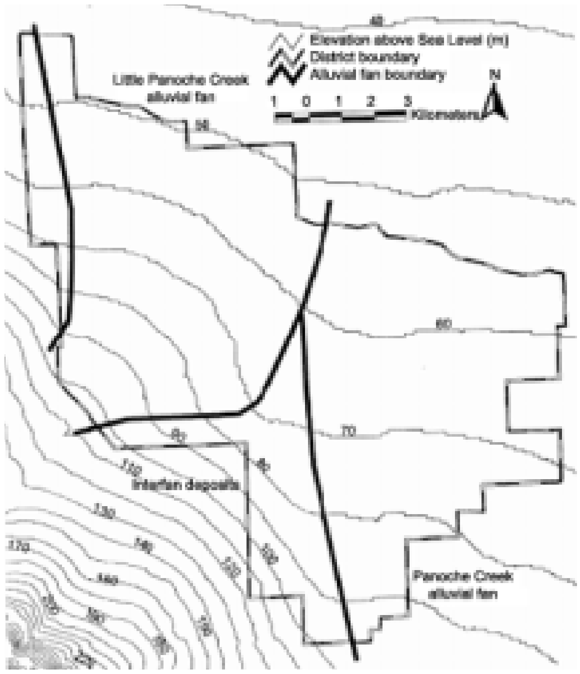
Environmental problems related to agricultural drainage water disposal on the western side of the San Joaquin Valley, California, the United States of America, have created a need for improved irrigation water and salt management (SJVDP, 1990). The presence of harmful trace elements, mainly selenium, in the drainage water is of major concern (Tanji *et al*., 1986) and has led to limitations on drainage water disposal to rivers and impoundments such as agricultural evaporation basins. Where water districts in the problem area fail to meet selenium and salt load targets, they risk monetary penalties and loss of access to disposal sites. Young and Wallender (2000a) raised the question of whether the constraints raised by current or future regulations will reduce drainage to the point where salt accumulation would occur and in which areas this might occur first. Second, they raised the question of the spatial distribution of drainage water disposal costing strategies. To answer these questions, they developed a methodology for the Panoche Water District to calculate the spatial distribution of water-, salt- and selenium-balance components using data collected by the water district. Furthermore, they developed and evaluated spatially distributed drainage water disposal costing strategies. The following is a brief overview of their research findings.

The Panoche Water District, situated in the western side of the San Joaquin Valley, is a typical example of a water district that needs to cope with disposal limitation in the form of selenium load targets. Figure 5 shows that the district lies on two alluvial fans and is generally flat with slopes of not more than 1 percent trending in a northeasterly direction.

The groundwater in the Little Panoche Creek alluvial fan contains sodium-chloride type water, relatively low in salt content, and with selenium concentrations ranging from 1 to 27 µg/litre (Young and Wallender, 2000b). In contrast, the groundwater in the Panoche Creek alluvial fan contains sodium-sulphate type water, relatively high in salt content, and with selenium concentrations ranging from 20 to 400 µg/litre. Due to overirrigation since the introduction of surface water delivery systems in the 1950s, the water table rose to within 1–3 m of the ground surface. Subsurface drainage was installed which maintained successfully the water table at an acceptable level for agriculture. However, due to irrigation and drainage practices the naturally occurring salts and selenium in the region's soils are mobilized and enter into the subsurface drainage system as well as into the shallow groundwater.

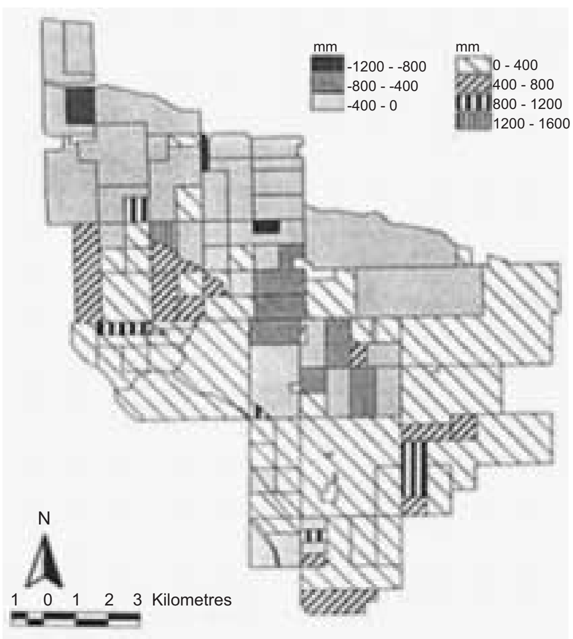
Calculation of a spatially distributed water balance revealed that downslope areas with shallow water tables receive groundwater discharge. Drains in these areas intercept lateral and vertical upward flowing groundwater, while the upslope undrained areas recharge to the groundwater (Figure 6). The highest salt load in the collected drainage water

FIGURE 5
Topography and boundaries of the Panoche Water District



Source: Young and Wallender, 2000a.

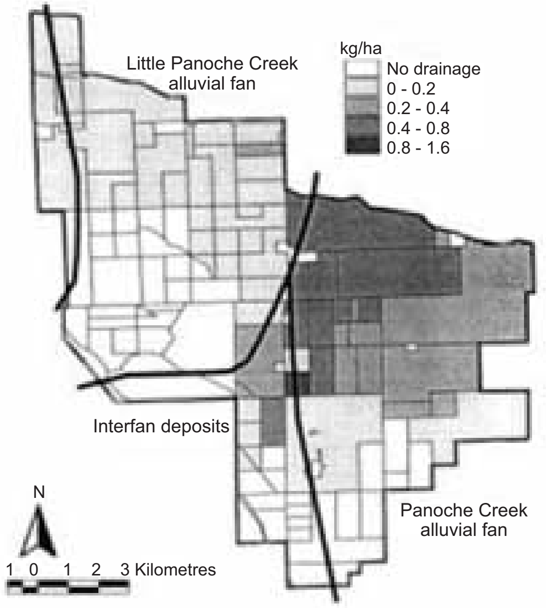
FIGURE 6
Annual recharge to groundwater in Panoche Water District



Source: Young and Wallender, 2000a.

occurred in the centre and northwestern part of the district that corresponds with the location of greatest drainage. Salts entered the drainage water via the groundwater with the maximum occurring at the alluvial fan boundaries. Accumulation of salinity occurred largely in the drained regions with the maximum occurring roughly in the regions of maximum groundwater recharge. In the undrained regions, more salt was removed from storage as compared to the drained regions, caused by greater deep percolation in the undrained areas coupled with salt pickup.

FIGURE 7
Annual amount of Se removed by the drains



Source: Young and Wallender, 2000b.

Figure 7 shows that the amount of selenium removed by the drains was greatest on the Panoche Creek alluvial fan and the interfan. The drainage system removed selenium through groundwater discharge. Selenium storage in the undrained areas decreased in proportion to the volume of deep percolation, while in drained areas selenium accumulated in areas similar to those where total salt storage increased.

Assuming a charge on drainage volume, spatial distribution of a drainage penalty per hectare would unfairly affect growers on the Little Panoche alluvial fan where relatively little selenium originates. In contrast, a charge levied per kilogram of selenium discharged from the drainage systems would result in growers on the Panoche Creek alluvial fan and the interfan paying more for drainage disposal in accordance with the higher selenium loads. Neither of the two methods of assessing drainage penalties addresses the poor water management in the upslope undrained areas that contribute to downslope drainage problems. A more equitable charge on the amount of selenium discharged into the environment from a control volume would account for excessive deep percolation as well as reflect differences in selenium loading caused by geological variations (Young and Wallender, 2000b).

THE USE OF MODELS IN RECOMMENDING SOLUTIONS AND ANTICIPATED RESULTS

Steps 4, 5 and 6 of the soft system methodology require models to predict changes as a result of a suggested implementation of measures and to enable decision-making.

Model characteristics

A major distinction is often made between simple and complicated models in which the former is frequently associated with engineering methods and the latter with scientific methods. The development of these different types of models and the use of the terms stem from the needs of various groups of professionals. Engineers, managers and decision-makers are in general looking for answers and criteria to base their management, decisions or designs on, while scientists are more interested in the underlying processes (Van der Molen, 1996).

The terms simple and complicated in relation to engineering or functional and scientific models are rather subjective. The distinction between scientific and functional refers not only to the purpose of modelling and the intended uses, but also implicitly to the approaches on which the models are based.

The three main groups of modelling approaches are mechanistic, empirical and conceptual approaches. Mechanistic, or as Woolhiser and Brakensiek (1982) define them, physically-based models are based on known fundamental physical processes and elementary laws. In groundwater modelling, this approach is also known as the Darcian approach. As this approach is based on elementary laws it should be, theoretically, valid under any given condition and therefore its transferability is extremely high. On the other hand, empirical approaches are based on relations that are established on an experimental basis and are normally only valid for the conditions under which they have been derived. Finally, conceptual calculation approaches are based on the modeller's understanding of fundamental physical processes and elementary laws, but these are not used as such to solve a problem. Instead, a concept of the reality is used to tackle a problem. The best-known example is the bucket-type approach to describe the flow of water through unsaturated soil.

Scientific models make use of mechanistic calculation approaches whenever possible and avoid the use of empirical and conceptual approaches. In contrast, functional models might include any of the three calculation approaches. Here, mechanistic approaches might be included as long as they do not conflict with other required model characteristics such as simplicity and short calculation time. Empirical and conceptual approaches are used in functional models as the only concern is that the model serves its intended purpose.

Regional models

The employment of a range of drainage water management options results in certain benefits, interactions and trade-offs not only in the place where the measure is implemented but also in adjacent and downstream areas. To enable decision-makers, managers and engineers to choose from different options and to study the effects of various alternatives, regional simulation models will need to be employed. Regional models normally include three main calculation modules, i.e. water flow and salt transport in the unsaturated or vadose zone, through the groundwater zone and through the irrigation and drainage conduits. Regional models normally require large amounts of data, and model calibration and validation is a time-consuming exercise. It is beyond the scope of this report to introduce the various models that have been developed and the reader may refer to Skaggs and Van Schilfgaarde (1999) and Ghadiri and Rose (1992) for more detail. The following sections introduce only some basic calculation considerations of water flow and salt transport in the unsaturated or vadose zone as these form the basis of several of the calculation methods presented in this publication. Where the water table in agricultural lands is controlled by subsurface drainage or where the water table is close to the rootzone, water and salt transport to and from the groundwater is considered as well. However, this publication introduces no specific groundwater models or calculation procedures for water and salt transport in the saturated zone.

Rootzone hydrosalinity models

Principles of rootzone hydrosalinity models

Rootzone hydrosalinity models may range from simple conceptual to complex scientific models. In the more simple models, the spatial component in the control volume (i.e. the crop rootzone) is typically assumed to be homogenous and space averaged (lumped), but water flow pathways are treated as distributed fluxes, e.g. deep percolation and rootwater extraction. The time increments taken may vary from irrigation intervals to crop growing season. Salinity is often treated as a conservative (non-reactive) parameter in simple models. The advantages of simpler functional models include more limited requirements for input data and model coefficients.

In contrast, the more complex, process-based rootzone models simulate water flow based on Richards' equation and treat salinity as a reactive state variable with simplified to comprehensive soil chemistry submodels. Such models provide greater understanding of the complexities in interactive physical and chemical processes. Complex scientific models require extensive input data and model coefficients, and carry out computations over small time and spatial scales.

A word statement of Richards' equation for the rootzone may be given by:

[Rate of change in soil water with respect to time]	=	[Rate of change in flux with respect to depth]	—	[Root water extraction sink with respect to depth and time]

In one dimension and taking small soil volume elements, Richards' equation is:

$$\frac{\partial q}{\partial t} = \frac{\partial}{\partial z}[K(q)\frac{\partial H}{\partial z}] - S_e(z,t) \qquad (1)$$

The terms in the parenthesis represent flux taken as the product of hydraulic conductivity (K) and hydraulic head gradient ($¶H/¶z$). Richards' equation is difficult to solve because there are two dependent variables volumetric water content (q) and H, the relationship is non-linear as K is a function of (q), and the water extraction sink (S_e) requires simulation of root growth by soil depth (z) and time (t). Once the soil water flow is simulated, the output data (q, flux) serves as input data for simulating soil chemistry.

Figure 8 describes some of the major chemical reactions involved in simulating changes in soil salinity.

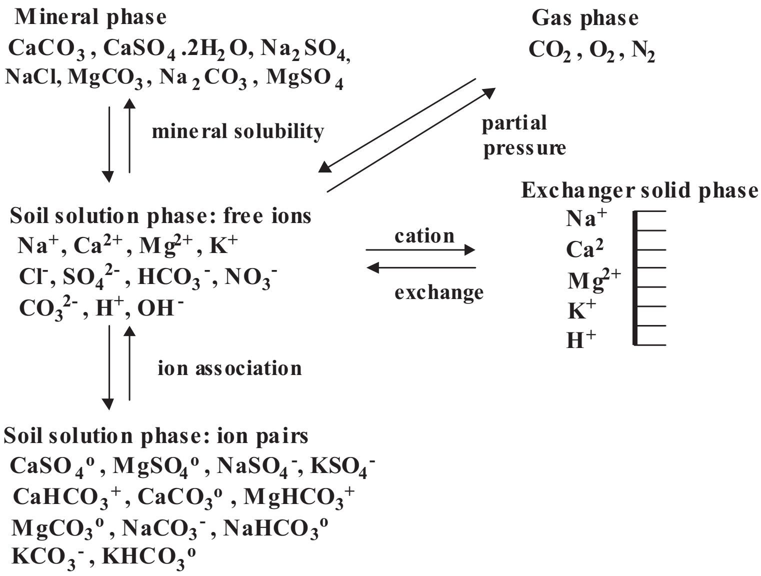
The reactivity and transport of chemical species is obtained from:

$$\frac{\partial q C_k}{\partial t} + r\frac{\partial \overline{C}_k}{\partial t} + r\frac{\partial \widehat{C}_k}{\partial t} = \frac{\partial}{\partial z}\left[qD\frac{\partial C_k}{\partial z} - qC_k\right] \quad k = 1,2,\ldots n \qquad (2)$$

The principal solute species ($C_{k=1..n}$) modelled are sodium (Na), calcium (Ca), magnesium (Mg), potassium (K), chloride (Cl), sulphate (SO_4), bicarbonate (HCO_3), carbonate (CO_3) and nitrate (NO_3). r is bulk density, D is dispersion coefficient, q is soil water flux, $\overline{C}_k$ is the exchangeable form and $\widehat{C}_k$ is the mineral form of the solute species.

An early hydrosalinity simulation model (Robbins *et al.*, 1980) was later extended to the widely used LEACHM (Wagenet and Huston, 1987). The US Salinity Laboratory has been active in modelling efforts for salt transport and major cations and anions such as UNSATCHEM (Simunek and Suarez, 1993; Simunek *et al.*, 1996) and HYDRUS (Simunek *et al.*, 1998, 1999), both in one and two dimensions. Trace elements of concern such as boron and selenium have yet to be incorporated into these models. Simultaneous water, solute and heat transport modelling of the soil-atmosphere-plant continuum has been developed at Wageningen Agricultural University, the Netherlands, in collaboration with ALTERRA (formerly the DLO Winand Staring Centre). The present version, SWAP 2.0, integrates water flow, solute transport and crop growth according to current modelling concepts and simulation techniques (Van Dam *et al.*, 1997).

FIGURE 8
Major chemical reactions in salt-affected soils



Source: Tanji, 1990.

Salt balance in the rootzone

The long-term sustainability of irrigated agriculture is heavily dependent on maintaining an adequate salt balance in the crop rootzone. For regions with a high water table, the salt balance needs to be expanded to include the shallow groundwater, too.

Kaddah and Rhoades (1976) examined salt balance in the rootzone subject to high water table with:

$$V_i * C_i + V_{gw} * C_{gw} + M_d + M_f = V_{dw} * C_{dw} + M_p + M_c + \Delta M_{ss} + \Delta M_{xc} \quad (3)$$

where:

V_i = volume of irrigation water (m^3);
V_{gw} = volume of groundwater (m^3);
V_{dw} = volume of drainage water (m^3);
C_i = salt concentration of irrigation water (kg m^{-3});
C_{gw} = salt concentration of groundwater (kg m^{-3});
C_{dw} = salt concentration of drainage water (kg m^{-3});
M_d = mass of salts dissolved from mineral weathering (kg);
M_f = mass of salts derived from fertilizers and amendments (kg);
M_p = mass of salts precipitated in soils (kg);
M_c = mass of salts removed by harvested crops (kg);
ΔM_{ss} = mass of changes in storage of soluble soil salts (kg); and
ΔM_{xc} = mass of changes in storage of exchangeable cations (kg).

Cation exchange between calcium, magnesium and sodium may modify the balance of these cations in the soil water and affect mineral solubility. Sodium minerals are more soluble than calcium minerals while magnesium minerals may range from highly soluble (sulphate type) to sparingly soluble (carbonate type).

Equation 3 contains some components that are not known or are small in relation to other quantities such as M_d, M_f, M_p and M_c (Bower *et al*., 1969). Moreover, the sources M_d and M_f tend to cancel the sinks M_p and M_c. If steady-state conditions are assumed for waterlogged soils, DM_{ss} and DM_{xc} may be assumed to be zero so that Equation 3 reduces to:

$$V_i * C_i + V_{gw} * C_{gw} = V_{dw} * C_{dw} \tag{4}$$

If the land is not waterlogged, $V_{gw} * C_{gw}$ drops out so that salt balance can be viewed simply and leads to such relationships as:

$$LF = \frac{V_{dw}}{V_i} = \frac{C_i}{C_{dw}} \tag{5}$$

Equation 5 expresses the leaching fraction (*LF*) and is the simplest form of the salt and water balance for the rootzone where surface runoff is ignored. Where the land is not waterlogged, V_{dw} consists of deep percolation from the rootzone.

Where the water table in agricultural lands is controlled by subsurface drainage, then the mass of salts in groundwater must be considered in Equation 4. Due to the nature of flow lines to subsurface drainage collector lines, the subsurface drainage collected and discharged is a mix of deep percolation from the rootzone and intercepted shallow groundwater. For example, for the Imperial Irrigation District, California, the United States of America, Kaddah and Rhoades (1976) estimated that deep percolation contributed 61 percent and shallow groundwater 39 percent to the tile drainage effluent based on chloride mass balance. They also estimated that tailwater contributed 10 percent to the total surface and subsurface drainage from the district. The ratio of tailwater plus intercepted deep percolation and shallow groundwater to applied water for the district was 0.36.

In the presence of high water table, shallow groundwater and its salts may move up into the rootzone (recharge) and down out of the rootzone (discharge) depending on the hydraulic head. Deficit irrigation under high water table may induce rootwater extraction of the shallow groundwater. The salinity level of the shallow groundwater is of some concern under such conditions. However, there does not appear to be a simple conceptual model of capillary rise of water and solutes. Chapter 5 and Annex 5 contain a method for computing capillary rise that requires extensive soil hydraulic parameters not normally available for field soils. Hence, the conceptual hydrosalinity models used in this paper are for the more simplified downward steady-state type.

Various versions of the salt balance, Equation 3, have served as the basis for numerous models including SALTMOD (Oosterbaan, 2001), CIRF (Aragüés *et al*., 1990) and SAHYSMOD, which is under preparation by the International Institute for Land Reclamation and Improvement, Wageningen, the Netherlands. Furthermore, the calculation methods as introduced by Van Hoorn and Van Alphen (1994) are based on the same concepts.

Chapter 3

Framework for selecting, evaluating and assessing the impact of drainage water management measures

DEFINITION OF DRAINAGE WATER MANAGEMENT AND TASKS INVOLVED

In the context of this publication, drainage water management refers to the management and control over the quantity and quality of the drainage water generated in an agricultural drainage basin in arid and semi-arid areas and its final safe disposal. This is achieved through irrigation water conservation measures and the reuse, disposal and treatment of drainage water. Managing drainage water at the field, irrigation-scheme and river-basin levels entails a number of activities including:

- regulating water table levels in the drainage system to ensure the maintenance of favourable soil moisture conditions for optimal crop growth and salinity control;
- developing irrigation and drainage water management strategies to ensure that disposal regulations and water quality standards are met and dealing with issues, problems and conflicts that might occur;
- setting distribution priorities and criteria for reuse in water scarce areas; and
- establishing cost sharing imposed on stakeholders for the use of poor quality water and required treatment to meet the water quality standards for drain discharge.

DRIVING FORCES BEHIND DRAINAGE WATER MANAGEMENT

The main reasons for developing a drainage water management strategy are: (i) prevention of economic and agricultural losses from waterlogging, salinization and water quality degradation; (ii) concern for quality degradation of shared water resources; and (iii) the need to conserve water for different water users under conditions of actual or projected water scarcity. In addition, the need to comply with drainage water policies and regulations can provide a strong incentive for improved drainage water management.

Many countries and states, e.g. Australia, India, Egypt and California, the United States of America, have a drainage water disposal policy and drainage effluent disposal regulations. In California, the United States of America, and in Australia, the drainage policy guidelines consist of difficult but achievable targets with an active enforcement of regulations. India and Egypt have policy guidelines of a general nature but have not reached the maturity of Californian and Australian laws. Law enforcement often fails in these countries, mainly as a result of administrative shortcomings and unrealistic quality guidelines for their conditions and resources. In these countries and in countries where clear laws and regulations are absent, the prevention of economic and agricultural losses and the conservation of water for other beneficial uses will normally be the main driving factors behind the development of a drainage water management strategy.

Physical drainage water management options

Figure 9 identifies the physical drainage water management options that are available to planners, decision-makers and engineers and how they relate to one another. The measures have been grouped into four categories: water conservation, drainage water reuse, drainage water treatment and drainage water disposal measures.

Conservation measures

A major goal of conservation measures is to reduce the volume of drainage effluent generated and the mass discharge of salts and other constituents of concern while at the same time saving

Figure 9
Physical drainage water management options and how they relate to one another

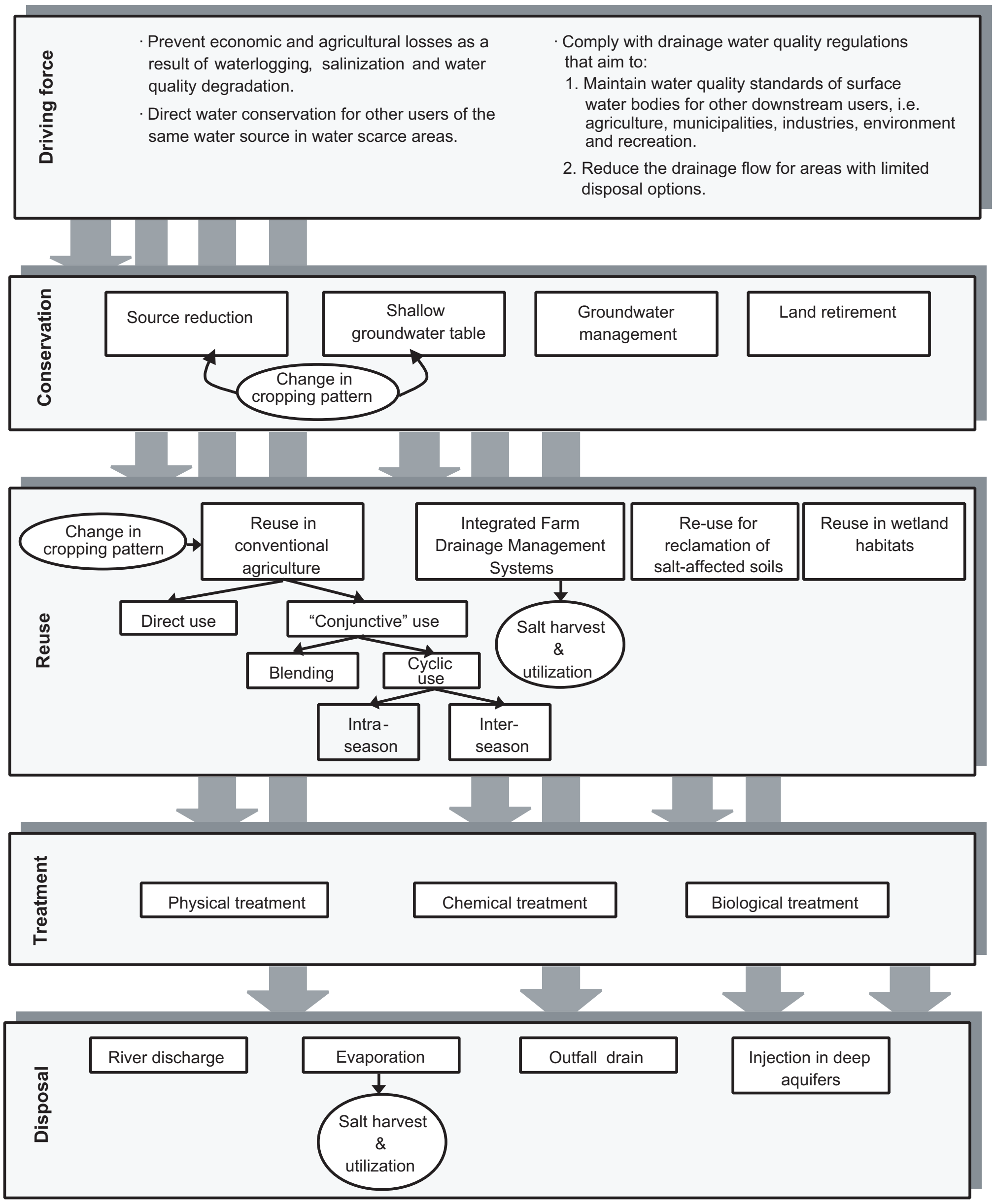


TABLE 2
Conservation measures, practices and points for consideration

Option	Practices	Points for consideration
Source reduction	Reduce the volume of deep percolation through: improving irrigation performance by surface irrigation, changing from surface to precision irrigation methods, modifying irrigation schedules, upgrading irrigation infrastructure, etc.	Farm- and system-level costs of system improvements have to be considered against the regional benefits. Salinization and concentration of toxic elements in the rootzone need to be prevented by guaranteeing minimum leaching.
Shallow groundwater table management	Encourage the use of shallow groundwater to meet crop evaporation through maintaining sufficiently high groundwater tables and practising deficit irrigation.	There is a danger of salinization and concentration of toxic elements in the upper soil layers due to induced capillary rise. Moreover, the risk of insufficient aeration during rainfall in the rootzone needs to be addressed.
Groundwater management	Pumping from vertical wells could control water tables. Pumped water of adequate quality could serve as a substitute for surface water.	Prevent overexploitation of groundwater resources. There is a danger of intrusion and upwelling of saline groundwater. As groundwater often contains elevated concentrations of salts and trace elements, their buildup to toxic levels and soil degradation needs to be prevented.
Land retirement	Retire or fallow irrigated lands that are heavily affected by waterlogging and salinity or those lands that generate drainage effluent with extremely high concentrations of salts and/or other trace elements.	Retired lands can become excessively salinized and high concentration of toxic elements can build up in the topsoil preventing natural vegetation from establishing itself. Contaminated bare lands can affect productivity and be a health risk to humans in the vicinity due to wind erosion.

Source: SJVDIP, 1999a, 1999c, 1999e, 1999f.

water for other beneficial uses. Conservation measures can directly affect the need for and extent of reuse as well as the quantity and quality of drainage effluent requiring disposal and/or treatment. Where competition for water quantity and quality among different groups of users is a major issue and where drainage effluent disposal is constrained (as in a closed drainage basin) or threatens ecologically sensitive areas, conservation measures are among the first to be considered. The various conservation measures include: source reduction (SJVDIP, 1999e); shallow groundwater table management (SJVDIP, 1999a and 1999f); groundwater management (SJVDIP, 1999f); and land retirement (SJVDIP, 1999c). Table 2 provides a short description of the four conservation measures and some points for consideration.

The first line of action in on-farm water conservation is source reduction or reducing deep percolation. Where the goal is not achievable by source reduction, it may be necessary to implement other conservation measures in combination with reuse measures. As different crops require different amounts of water for optimal crop growth and as rooting patterns differ among crops, source reduction and water table management could be combined with a change in cropping pattern to optimize the desired effect of the measures. Changes in cropping pattern should only be considered where compatible with the broader development objectives for the area under consideration.

Reuse measures

The major aim of reuse measures is to reduce the amount of drainage effluent while at the same time making additional water available for irrigation and other purposes. Reuse measures comprise: reuse in conventional agriculture; reuse in saline agriculture; Integrated Farm Drainage Management (IFDM) systems; reuse in wildlife habitats, wetlands and pastures; and reuse for

TABLE 3
Reuse measures, practices and points for consideration

Option	Practices	Points for consideration
Reuse in conventional agriculture	Agricultural drainage water is collected and redistributed among farmers. Reuse can be direct or in conjunction with other sources of irrigation water. Conjunctive use can be through blending or by cyclic use of drainage water and other sources of irrigation water.	The extent of reuse depends on the quality of the drainage effluent, time of availability, crop tolerance, etc. Soil quality degradation and production losses need to be prevented through mitigation measures. Residuals of reused waters, which are often highly concentrated with a reduced volume, need to be managed.
Reuse in saline agriculture	Moderately and highly saline drainage water is collected and used to cultivate salt tolerant shrubs, trees and halophytes.	Sustainability of the saline agricultural systems needs to be safeguarded while the highly concentrated drainage effluent needs to be managed.
IFDM System	On-farm sequential reuse of agricultural drainage water on crops, trees and halophytes with increasing salt tolerance. In every reuse cycle the volume of drainage water decreases while the salt concentration increases. The final brine is disposed in a solar evaporator. Salt utilization might be feasible.	To maintain adequate control of soil salinity and sodicity and to prevent a buildup of toxic elements, the leaching fraction must be sufficient. The solar evaporator should be designed in such a manner that it will not sustain aquatic life and attract birds. The salts have to be disposed of in a safe and sustainable manner.
Reuse in wildlife habitats and wetlands	Where of suitable quality, drainage water may be utilized to support wildlife, including waterbirds, fish, mammals and aquatic vegetation that serves as food and cover for wildlife.	Of primary concern is the possible presence of trace elements that may be toxic to wildlife through bioaccumulation in the food chain e.g. selenium, molybdenum and mercury.
Reuse for reclamation of salt-affected soils	Use of (moderately) saline drainage water for initial reclamation of saline, saline sodic or sodic soils. On sodic soils the saline water may help prevent soil dispersion and degradation of soil structure.	Once initial reclamation is obtained, water of sufficient quality and quantity needs to be available for desired land use. Initially, the drainage effluent will be highly saline and/or sodic and needs to be disposed of in a safe manner.

Source: SJVDIP, 1999a.

initial reclamation of salt-affected lands (SJVDIP, 1999a). Table 3 provides a brief description of the alternative reuse measures.

Reuse measures can be implemented in combination with conservation measures. Where the drainage water is of relatively good quality, its reuse potential in conventional agriculture is high. Where it is moderately to highly saline, its reuse may be limited to salt tolerant plants. In California, the United States of America, a new IFDM system reuses drainage water sequentially in high water table lands with no opportunities for disposal of subsurface drainage (SJVDIP, 1999a). Freshwater is used to grow salt sensitive crops, and subsurface drainage water from it is reused to grow salt tolerant crops. Drainage water from the salt tolerant cropland is used to irrigate salt tolerant grasses and halophytes. When the drainage water is no longer usable, it is disposed into solar evaporators for salt harvest. In this system and others, it is always necessary to generate a minimum volume of drainage effluent to prevent the rootzone from becoming too contaminated for any beneficial water use activity. Similar systems are being developed and tested in Australia.

Treatment measures

Drainage water treatment in a drainage water management plan normally takes place only under severe constraints, such as stringent regulations on disposal of saline drainage waters into streams, or severe water shortage. The drainage water treatment options are based on physical, chemical and/or biological processes (SJVDIP, 1999b). Many of these processes are borrowed

TABLE 4
Array of drainage water treatment options

Description	Processes	Constituents removed or treated
Physical / chemical	Sedimentation / coagulation	Remove sediments and associated nutrients, pesticides and trace elements in sedimentation ponds with or without coagulants.
	Adsorption	Remove soluble constituents onto surfaces of adsorbents.
	Ion exchange	Exchange constituents with another using ion exchanger resins or columns.
	Reverse osmosis	Under pressure, separate out dissolved mineral salts through semi-permeable membranes.
	Coagulation / precipitation	Use chemicals such as alum to coagulate or precipitate constituents of concern.
Biological	Reduction / oxidation	Reduce oxidized mobile forms such as selenate to reduced immobile forms such as elemental selenium through microbially mediated processes.
	Volatilization	Some plants and microbes are capable of taking up constituents such as selenium and volatilizing the methylated forms into the atmosphere.
	Plant / algal uptake	Certain terrestrial plants and algae are capable of extracting large amounts of constituents such as selenium, nitrate and molybdenum.
	Constructed flow-through wetlands	Constituents such as selenium and heavy metals are removed from drainage water. For selenium, the principal removal mechanism is reduction to elemental selenium and organic forms in the detrital matter. For heavy metals, the principal sink mechanism is sorption or fixation on the sediments.

Source: SJVDIP, 1999b.

from water treatment processes for drinking-water, sewage and industrial wastewater. A few are new processes to remove selenium. The water quality requirements of the treated water need to be well understood prior to the selection of any treatment measure. Table 4 provides the array of drainage water treatment options.

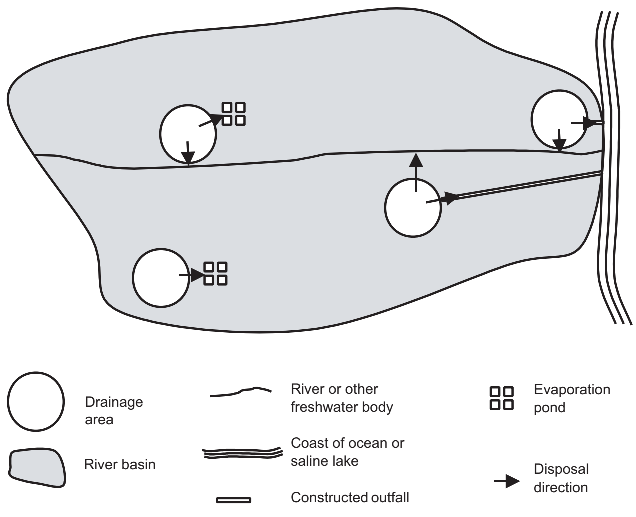
The high costs of many of the treatment processes make them unsuitable for agricultural reuse. Treatment measures such as reverse osmosis are normally only considered for high-value purposes such as drinking-water supply. One option is to partially treat saline waters to a level that could be used for agriculture (e.g. desalt to about 1 dS/m instead of less than 0.1 dS/m). However, the cost of brine management and disposal remains a major problem.

An exception to the typical high cost is the treatment of drainage water through constructed wetlands. Investigations are underway in California, the United States of America, in which wetland cells are planted with cattails, saltmarsh bulrush, tule, baltic rush, saltgrass, smooth cord grass and rabbitsfoot grass. The cells remove about 60–80 percent of the mass of selenium with hydraulic residence times ranging from 7 to 21 days. This treated water is then disposed into agricultural evaporation ponds with reduced toxic selenium effects on waterbirds. Constructed wetlands appear to have some potential to protect aquatic ecosystems and fisheries either downstream or in closed basins.

Disposal measures

Even after the successful implementation of conservation and reuse measures, there will always be a residual volume of drainage effluent requiring disposal. Disposal options depend mainly on the situation of the drainage outlet in relation to natural disposal sites such as rivers, streams, lakes and oceans. Disposal options to surface water bodies comprise discharge into rivers and streams, river mouths and constructed outfall drains for direct disposal to oceans or saline lakes, and evaporation ponds (Figure 10).

Figure 10
Options for disposal to surface water bodies



Another potential disposal option is deep-well injection. However, in California, this approach failed due to the slow permeability of the geologic strata from the plugging of conducting pores (Johnston *et al*., 1997). Besides the natural drainage conditions, the suitability of each of these measures depends on: the quality and quantity of drainage water requiring disposal; environmental and health risks; available technology and resources; and economic considerations. Table 5 provides a short description of the disposal measures.

Drainage water disposal into natural surface water bodies should entail minimal deleterious impacts on other downstream water uses (agricultural, industrial and municipal) and wildlife. Disposal on land in closed basins and agricultural evaporation basins should avoid undue harm to the ecology, particularly aquatic biota including fish and waterbirds.

Table 5
Disposal measures, practices and points for consideration

Option	Practices	Points for consideration
River discharge	The drainage effluent is released into rivers, streams, etc. from where it finds its way to an ultimate salt sink, i.e. oceans and salt lakes. This option is especially appropriate during high discharge periods.	Disposal in rivers and lakes should not unduly impair other downstream uses including water required for sustaining fragile aquatic ecosystems. River disposal is often limited by disposal regulations.
Evaporation ponds	Disposal of drainage water in natural depressions or specially designed unlined basins. The impounded water dissipates through evaporation and inadvertent seepage losses, and deposits salts and trace elements.	Concentration of trace elements could adversely affect birds and wildlife through bioconcentration in the aquatic food chain to toxic levels. Adverse toxic impacts should be mitigated through special measures. Excessive seepage losses may pose a serious contamination risk to groundwater resources.
Outfall to saline lakes or oceans	Constructed main drain or disposal into river mouths to discharge effluents into the ocean or saline lakes.	Construction of outfall drains over long distances is normally an expensive undertaking and should only be considered where other alternatives are not feasible or the stream water quality is fragile. Disposal into oceans, bays and estuaries may be restricted if toxic trace elements are present. Disposal into tidal rivers needs tidal gates to prevent saltwater intrusion during high tide.
Deep-well injection	Treated drainage water is often injected into deep underlying permeable substratum in confined aquifers.	Formation of microbial slimes and colloidal particles may affect permeability in the stratum. There is a risk of seepage of poor quality water into fresh groundwater bodies.

Source: SJVDIP, 1999d, 1999g.

Non-physical drainage water management options

In order to be fully effective, non-physical management measures, i.e. policy and legislation, should accompany physical drainage water management options. Different countries have implemented several policies, and these are often part of a general pollution policy.

One of the most frequently mentioned principles with respect to pollution is that of the polluter pays. This means that economic policy instruments are applied to change the behaviour of farmers in such a way that pollution is minimized or that at least the polluter pays for its effects. Weersink and Livernois (1996) provide an overview of such economic policy instruments for resolving water quality problems from agriculture. This overview was compiled for pollution by nutrients in humid temperate areas, such as Canada. However, some of these policies have relevance to pollution from the drainage of irrigated lands. The main problem of policy instruments in drainage water management is that water quality degradation from agriculture is non-point source pollution. Direct links between agricultural practices and water quality degradation are often difficult to determine and quantify.

Weersink and Livernois divided the policy instruments into two groups: (i) those based on the performance of an agricultural system, i.e. on the actual amount of pollution that caused; and (ii) those based on agricultural practices such as fertilizer and agrochemical uses (Table 6). In the latter case, the underlying assumption is that certain agricultural practices will lead to more or less pollution. These policy instruments try to influence the practices and, indirectly, the pollution resulting from these practices.

Table 6
Economic policy instruments

Base	Target	Form
Performance of agricultural systems	Emission	Charges
	Ambient	Charges
		Salinity permits
Agricultural practices	Inputs	Charges
		Subsidies
	Outputs	Charges
		Subsidies

Source: based on Weersink and Livernois (1996).

Emission levels

Fees are levied on the discharge of the polluter into the water. The aim is to stimulate the polluter to adopt practices that minimize pollution or to make the polluter pay for some of the damage caused. This requires measurement of individual discharges, which is often costly and not always practical to implement. Measurements at the individual farm level are probably only feasible with modern and large farms, such as in the United States of America and Spain. However, the case of the Panoche Fan (Chapter 2) shows that it remains difficult to establish equitable fines. Discharges from upstream farms with natural internal drainage cannot be measured while they may contribute significantly to the pollution in downstream regions. An alternative to measuring and levying fees on individual farms is to implement it at district level or at that of a water users association. This has the disadvantage that farmers applying conservation measures may still pay for the pollution caused by badly managed farms in the same district.

When basing fines on emission levels, there should be a decision as to whether to base them on concentrations of polluting elements or on the total load (i.e. concentration multiplied by flow). Concentration is the water quality parameter used in drinking-water standards or for the health of aquatic animals and plants. Placing only concentration limits on discharge might encourage dilution or inefficient water use. Where both concentration and load limits are enforced, they tend to promote efficient water use.

Ambient levels

Basing charges on the pollution in the downstream receiving water body can sometimes be a feasible alternative to basing them on the emission level, i.e. where the source of pollution is extremely difficult to measure. For example, in the case of pollution of groundwater, individual contributions to the pollution are difficult to measure and thus difficult to quantify. A disadvantage of using ambient levels under these conditions is that all farmers, including those applying sound water management practices, are charged equally.

Another case where this policy measure may be an attractive option is where the carrying capacity of the receiving water body changes continuously. For example, in India during the monsoon, river discharge is high and effluent disposals during this period increase downstream concentrations only slightly. On the other hand, during the dry season when river flow is low, emissions affect water quality substantially. In this case, using the ambient criterion reflects the major concern: maintaining river water quality for downstream users. This method stimulates corrective measures during critical periods.

Salinity permits

An interesting solution for salinity management on a regional level is that adopted in the Murray-Darling basin in Australia. States within the basin have to meet electrical conductivity (*EC*) levels at the end of their river valleys, this in order to maintain a favourable water quality in the entire downstream river. In order to reach this goal, a system of salt credits and debits is used. Credits are obtained for the implementation of any works that reduce the salinity in shared rivers. Debits are incurred based on the estimated shortfall in protecting shared rivers. The balance of credits and debits is registered for each state, and as a general principle each state must be in credit. The credits and debits are converted to *EC* impact at a location in the downstream area of the basin. This method allows states and catchment management authorities to decide on the most cost-effective options for their area whilst contributing to the overall basin-wide river salinity management plan (Murray-Darling Basin Ministerial Council, 2001).

Charges on inputs

Instead of charging for pollution outflow from irrigation systems, economic incentives can be directed at reducing the inflow, which will normally lead to a reduced pollution outflow. Thus, as a way of reducing pollution, input charges seek to conserve water and minimize the use of agrochemicals and fertilizers. Volumetric water pricing is the main way of providing an economic incentive to save water. However, research from Iran (Perry, 2001) has highlighted some limitations of this approach. In the Iranian case, to serve as a proper economic incentive to change to pressurized irrigation, the total costs of water would need to equate to 60 percent of the revenues of a wheat crop, while at the moment they stand at 5 percent. Such an increase is probably politically impossible and would reduce farm incomes drastically. Surface irrigation improvement is less costly than changing to pressurized irrigation. Therefore, volumetric water pricing may be an economic incentive to improve surface irrigation water management.

Charges on inputs should also consider the way in which farmers can reduce water consumption. The main changes farmers can make are: grow less-water-demanding crops; apply water saving technologies; and change over to rainfed farming (generally not practised in arid and semi-arid areas). All these responses lead to less water consumption, but sometimes also to significant drops in farm income level.

On the basis of computer simulations, Varela-Ortega *et al*. (1998) analysed the responses of farmers in Spain to increasing water prices. Results showed that in modern irrigation districts, where irrigation efficiency is already quite high, no changes in water demand would occur unless the prices became restrictively high and farmers needed to change to rainfed farming. In contrast, in areas with the older irrigation systems, farmers would have ample scope for improving their water use efficiency.

An alternative to volumetric pricing is tiered water pricing. In this system, farmers pay a lower volumetric price for a reasonably necessary amount of water to grow a certain crop. A higher volumetric price is charged for additional amounts of water. Such a system is explicitly designed to reduce the amounts of water that generate most of the drainage water. The reasonably necessary amount should include the minimal leaching requirement in order to maintain a favourable salt balance in the rootzone unless sufficient rainfall occurs in the area. A main problem would be to define the 'reasonable necessary amount of water'. In Broadview Water District, California, the United States of America, tiering levels differ only between crops and not between soils, because farmers consider the latter approach to be inequitable. Tiering levels were established at 90 percent of the average depths applied in the years before the system was implemented. This system of tiered water pricing led to a decrease in water applied to five of the seven main crops. More importantly, the drained volume in 20 of the 25 subsurface systems in the water district decreased by 23 percent and the salt load declined by 25 percent (Wichelns, 1991).

Volumetric pricing policies are typically only applicable in on-demand systems. Other systems deliver fixed amounts of water to farms, and they fall outside the responsibility of the individual farmers. In such systems, more gains could be obtained by matching supply and demand more closely to each other in a technical/operational way.

Subsidies on practices

A changeover to water saving or pollution prevention measures might not always be the objective of individual farmers nor might it always be to their advantage. In the case of Iran, it was calculated that the investments in sprinkler irrigation are of the same order of magnitude as the value of the water that can be saved, if this is used to irrigate a larger area or high-value crops (Perry, 2001). However, if the water saved is diverted to irrigate land in other areas, upstream farmers will be investing while benefits accrue to other areas. Another example is the case of farmers located in the higher areas with free drainage in the Panoche Fan. Farmers in these areas contribute much to the pollution, but there is no direct incentive for them to invest in water saving or pollution prevention measures. In both cases, subsidies might stimulate farmers to invest in water saving technologies or pollution prevention measures.

Charging/subsidizing outputs

A final instrument proposed by Weersink and Livernois (1996) is a charge on the crops. In the case of nitrogen leaching, the charge focuses on crops that need large amounts of nitrogen, and thus have a higher risk of causing pollution. In irrigated agriculture, one could think of policies that discourage growing crops with large water requirements (such as alfalfa) by charging the crops or by promoting deep-rooting crops or less-water-demanding crops through subsidies. A common example is where farmers pay a price per area depending on the crops grown.

Combined measures

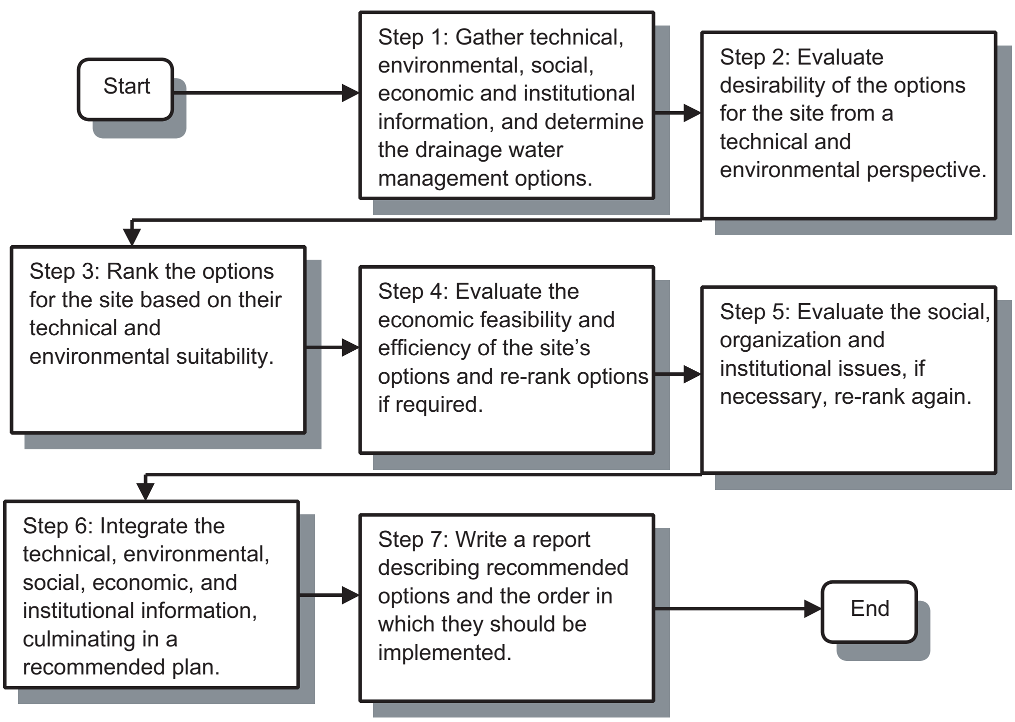
A single economic or policy instrument will probably not resolve water pollution problems originating from irrigated agriculture. Combinations of policies can help to enforce their effect. Especially in situations where farmers are inequitably charged, a combination of measures might help to charge farmers more equitably. For example, emissions from an irrigation district could be measured and fines applied where limits are surpassed. These fines could then be distributed amongst the farmers, applying a ranking based on how farmers have used their inputs or what measures they implement. The most inefficient users should then pay comparatively more than the most efficient ones.

Selection and evaluation of drainage water management options

Normally, more than one drainage water management option will need to be considered and implemented to attain the desired objectives. In this case, interactions and trade-offs occur. Because of the complexity of the processes and interactions, computer models are best suited to selecting an optimal combination of drainage measures. As was explained in Chapter 2, the selection of a set of drainage water management measures requires more considerations than solely technical feasibility. Figure 11 shows a flowchart for selecting a set of drainage water management measures (AHCC, 2000).

Step 1 is to gather technical, environmental, social, economic and institutional information about the site and the available drainage water management options. Step 2 is to evaluate the desirability of the options from a technical and environmental perspective. Typically, this step

Figure 11
Flowchart of process for selecting an optimal set of drainage water management options



Source: based on AHCC, 2000.

requires the use of computer models. Figure 9 might also help in the evaluation process. Step 3 involves ranking the site's options, based on the technical and environmental desirability. Step 4 involves an analysis of the relative economic efficiency of the various options. Marginal cost curves may be useful in determining which options are more efficient than others. When evaluating the economic efficiency of a highly complex, multi-parameter system, it may be advantageous to turn to a computerized economic optimization model. The course of action ultimately decided upon also depends on the social, organizational and institutional suitability of the proposed options and may run counter to what is economically efficient. Step 5 evaluates the social and institutional issues. Step 6 is perhaps the most difficult as it always involves intuitive judgement and a certain measure of creativity. Collating all the information and developing a stepwise path for implementing the favoured options takes time and considerable thought. The preparing of a report describing the recommended options and their order of implementation (Step 7) follows directly from Step 6.

BENCHMARKING

The Australian National Committee on Irrigation and Drainage is exploring another approach called benchmarking as part of its efforts to improve the performance of irrigated agriculture. The principal goal of benchmarking is to "find and implement best practice for the organization in question" by "learning about their own organization through comparison with their historical performance and with practices and outcomes of others" (Alexander, 2001). In 1998/1999, this committee evaluated 47 parameters in 46 water providers to gauge system operation, environmental issues, business processes and financial performance. New parameters to be considered with specific regard to on-farm water use include: security of water supply, water savings from operational and seepage remediation, metering of supplies, water trading, salt balance, water quality monitoring, and operating costs per unit length of channel or pipeline. The benchmarking reports and compiled data have already led the water industry to consider and/or adopt improved technologies to enhance economic, environmental and social performance. This approach has received positive responses from the water industry, and the plan is to now evaluate additional comparative parameters, such as groundwater quality, absolute changes in water table, extent of water reuse and recycling, average water use on major crops, and financial reports on economic value added.

Chapter 4

Water quality concerns in drainage water management

INTRODUCTION

There are several factors to consider when determining the opportunities for and constraints on the safe use, treatment and disposal of agricultural drainage water. Information and data desired at the site of drainage water production include: rate of drainage water production per unit area, chemical concentration of constituents of concern, and the rates of mass emission. Drainage water management requires additional information and data on drainage water quality and its suitability for the intended water uses as well as an understanding of environmental and health concerns. Upstream drainage water management affects the needs and water quality requirements of downstream water users. As reliable references are already available on estimating drainage water production volumes (Smedema and Rycroft, 1988; Skaggs and Van Schilfgaarde, 1999), this chapter concentrates on the main factors affecting drainage water quality and water quality needs and concerns for other water users and incorporates previous work done by FAO (1997c).

DRAINAGE WATER QUALITY

Table 7 provides a summary of changes in water quality expected in irrigation return flow relative to irrigation water applied (Tanji *et al.*, 1977). The expected differences in quality in the return flow are described relative to the supply water because actual concentrations in supply waters vary. The operational spill waters (bypass water) from distribution conveyances are not expected to differ much from the quality of the supply water except for some pickup or deposition of sediments. In contrast, surface runoff or irrigation tailwater tends to pick up considerable amounts of sediments and associated nutrients, phosphorus in particular, as well as water-applied agricultural chemicals such as pesticides and nitrogen fertilizers (especially anhydrous ammonia). Tailwater is typically similar to the applied waters in salinity and oxygen demanding organics, termed biochemical oxygen

TABLE 7
Expected quality characteristics of irrigation return flow as related to applied irrigation waters

Quality parameters	Operational spills	Irrigation tailwater	Subsurface drainage
General quality	0	+	++
Salinity	0	0, +	++
Nitrogen	0	0, +, ++	++, +
Oxygen demanding organics	0	+, 0	0, -, --
Sediments	+, -	++	--
Pesticide residues	0	++	0, -, +
Phosphorus	0, +	++	0, -, +

0 not expected to be much different than the supply water.
+, - slight increase/pickup or decrease/deposition expected.
++ expected to be significantly higher due to concentrating effects, application of agricultural chemicals, erosional losses, pickup of natural geochemical sources, etc.
-- expected to be significantly lower due to filtration, fixation, microbiological degradation, etc.
Source: Tanji *et al.*, 1977.

demand (*BOD*). Subsurface drainage is enriched in soluble components such as dissolved mineral salts and nitrates, very low in sediments, whereas other quality parameters are similar to the irrigation water. These changes in water quality of irrigation return flow depend on a number of factors including irrigation application methods, soil properties and conditions, application of agricultural chemicals, hydrogeology, drainage system, climate, and farmers' water management.

In many regions of the world, municipalities and industries discharge wastewater into open drains initially intended for the conveyance of only agricultural drainage and storm water. In developing countries especially, municipal and industrial wastewater is often insufficiently treated before disposal into such open drains. The result is a risk that agricultural drainage water quality might be seriously contaminated with microbes, pathogens, toxic organics and trace elements including heavy metals.

A knowledge of the composition of the drainage effluent and the ability to predict changes in the composition as a result of changes in crop, irrigation or drainage water management practices are important in the planning and management of drainage water.

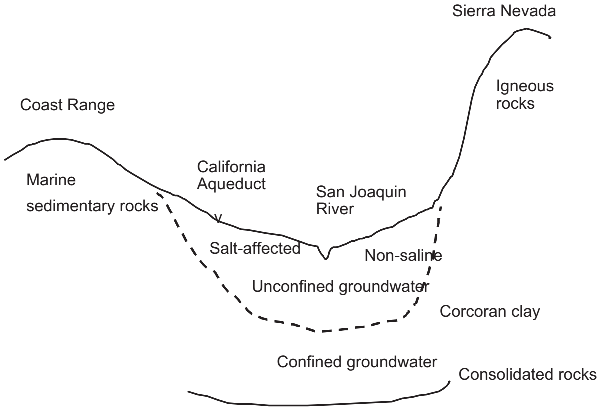
FACTORS AFFECTING DRAINAGE WATER QUALITY

Geology and hydrology

The geology of the region plays an important role in drainage water quality. Through weathering processes, the types of rocks (both primary and sedimentary) in the upper and lower strata define the types and quantities of soluble constituents found in the irrigated area. The oceans have submerged many parts of the continents during a period in their geological history. The uplift of these submerged geological formations and receding seas have left marine evaporites and sedimentary rocks behind, high in sea salts including sodium, chloride, magnesium, sulphate and boron. These geological formations exist in varying thicknesses, depths and extents on the continents. Through hydrological processes, solutes can enter the upper stratum by irrigation or floodwater, upward groundwater flow in seepage zones, with rising groundwater levels, or capillary rise. Once the solutes are in the upper strata, they influence the quality of agricultural drainage water through farmers' irrigation and drainage water management. The following example shows how the geology and hydrology of an area influence the quality of agricultural drainage water. It also illustrates the relationship between geomorphology, waterlogging and salinization.

Figure 12 shows a schematic cross-section of the San Joaquin Valley with the San Joaquin River as the principal drainage course for this river basin. The eastern side of the valley was formed from the alluvium of the Sierra Nevada, which consists mainly of granitic rocks. The soils derived from Sierran alluvium tend to be coarse textured and non-saline. The eastern groundwaters are characterized as low-salt calcium-bicarbonate-type water with total dissolved solids (*TDS*) typically in the 200-500 mg/litre range. In contrast, the soils on the western side were formed

FIGURE 12
Cross-section of the San Joaquin Valley



from alluvium of the Coast Range made up of uplifted marine sedimentary rocks. The soils on the western side tend to be finer textured and saline. The groundwaters on the western side are characterized as moderately saline sodium-sulphate-type waters with *TDS* typically in the 1 000-10 000 mg/litre range. The unconfined aquifer in both sides of the valley is gradually being filled up with decades of irrigation deep percolation. The soils in the valley and lowest part of the alluvial fans in the western side are waterlogged and salt affected. A nearly water-impermeable clay layer known as the Corcoran clay, about 200 m deep, serves as the boundary between the unconfined and confined aquifer. The groundwaters in the confined aquifer contain from 500 to 1 000 mg/litre *TDS*. During the geologic past, plate tectonics caused the horizontal-lying Corcoran clay in the shallow sea to tilt upwards forming the Coast Range.

FIGURE 13
Freebody diagram of water flows in the San Joaquin Valley

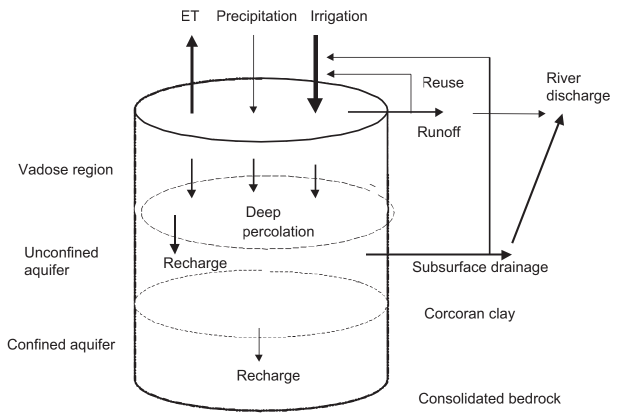


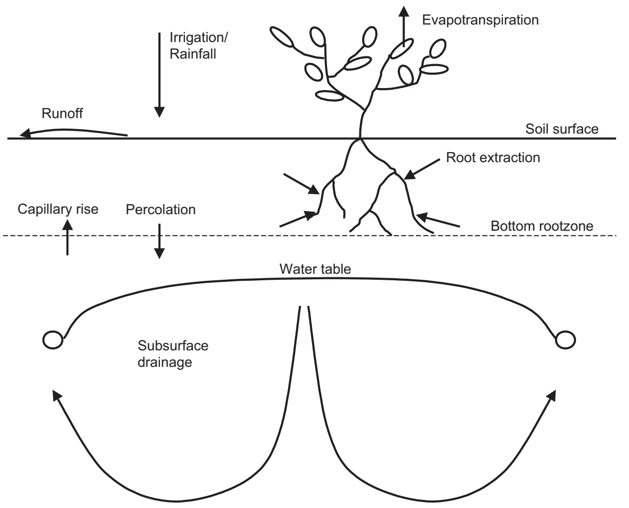
Figure 13 is a freebody diagram of the waterlogged irrigated lands on the western side showing the water flow pathways in the surface and subsurface. The applied irrigation water is about 450 mg/litre calcium-bicarbonate-type imported water from the Sacramento River basin to the north. Much of the surface runoff is captured and reused on site. Much of the collected saline subsurface drainage water (4 000-10 000 mg/litre *TDS*, sodium-sulphate type) is discharged into the San Joaquin River. Especially high concentrations of trace elements such as boron and selenium originating from the marine sedimentary rocks, found in the subsurface drainage waters, have given rise to environmental and health concerns. Discharges from these areas are now constrained by waste discharge requirements.

A second example comes from the Aral Sea Basin, where in total 137 million tonnes of salt are annually discharged, of which 81 million tonnes (59 percent) originate from the irrigation water and 56 million tonnes (41 percent) from the mobilization of salts from the subsoil (World Bank, 1996). In the mid-stream areas, mobilization of subsoil salts is the most substantial. The annual discharge from the Karshi oblast (Uzbekistan) is 10.8 million tonnes of salts, which corresponds to about 34 tonnes per hectare. About 4.3 million tonnes (about 40 percent) originate from irrigation water and the remainder are mobilized salts from the subsoil through irrigation and drainage (World Bank, 1998). Further, in Australia, large amounts of salts are added to the soil profile by atmospheric deposition of salts from upwind wind erosion of salt pans.

Soils

Figure 14 depicts the water movement over the soil surface and through the soil profile. Soils serve not only as a medium for plant growth but also store water and nutrients and serve as the porous transport media. The soil's eroding capacity and chemical weathering leads to the generation of water-borne suspended particles and solutes, ranging from nutrients to all kinds of contaminants (Van Dam *et al*., 1997). Therefore, to understand the role of soils on drainage water quality, it is necessary to understand water movement over and through the soil and the associated suspended and dissolved substances it carries.

Figure 14
Water flow over and through the soil



Of the water added to the soil, either in the form of rainfall or irrigation, part is lost through runoff and direct evaporation at the soil surface. Runoff water collects in natural and constructed surface drains from where it finds its way to the final disposal site (a river, evaporation pond or outfall drain to the ocean or saline lake). The other part infiltrates into the soil. This water fills up the soil pores and restores the soil moisture content up to field capacity under free drainage. The stored water is now available for plant root extraction to satisfy the water requirement of the crop. Any water in excess of field capacity percolates below the rootzone to greater depth in the vadose zone. The deep percolation water may eventually serve as recharge to the groundwater or saturated zone. In irrigated areas with shallow groundwater tables, the recharge is immediate and causes the water table to rise. Where subsurface drainage is installed in waterlogged soils, the drainage system removes deep percolation and groundwater. Where the soil moisture content in the rootzone drops as a result of evapotranspiration and if there is no recharge from irrigation or rainfall, capillary rise into the rootzone might occur, depending on the water table depth, soil texture and structure, and seepage.

Runoff in irrigated agriculture is mainly related to the intensity of irrigation and rainfall events in comparison to the infiltration capacity of the soil. Where the infiltration rate is smaller than the irrigation or rainfall intensity, water will accumulate on the soil surface and run off under a minimum surface slope. Soil degradation, in terms of compaction and crust formation as well as cultivation on steep slopes, promotes surface runoff. Through the physical forces of the running water, soil particles become suspended in the water and are transported to open drains, ditches, streams, rivers and lakes. Deposition of suspended sediments may occur downstream when current velocities decrease. Suspended soil particles are harmful to aquatic life as they diminish light transmission, but also because chemical contaminants may be associated with suspended sediments (NRCS, 1997). Degradation of drainage water quality as affected by runoff from agricultural land is especially important in hilly areas and in areas with excess rainfall.

In the more arid areas and in flat plains, water flowing through the soil profile and associated solute fluxes affected by mineral solubility and adsorption processes are of more importance for the final quality of the drainage effluent than surface runoff. In some places, weathering of soil particles might play a major role in the quality of drainage water (such as dissolution of gypsum). However, in general the soil's ability to adsorb and release through ion exchange and transform chemical elements through microbially-mediated redox reactions plays a more important role. In this context, soil characteristics such as water holding capacity, hydraulic conductivity, clay and organic matter content, soil minerals and soil microbes are important characteristics. Soil degradation in the form of erosion, compaction and loss of biological activity reduces the water and solute holding capacity of the soils. This increases the mobility of solutes through the soil and increases the risk that pollutants such as salts, nutrients and pesticides will be lost both to groundwater and through interception by subsurface drainage to surface water (NRCS, 1997).

Climate

As the major transport of solutes through the soil is by the movement of water, climate plays a major role in determining drainage water quality. In humid tropics and temperate regions, the dominant movement of water through the soil is vertically downwards. Solutes, which are brought onto the soil by farmers or are naturally present in the upper soil layers, are leached into deeper soil layers and groundwater. Conversely, in arid climates where evaporation largely exceeds precipitation the dominant water movement through the soil is vertically upwards except during rainfall or irrigation events. Therefore, the chemical composition of deeper soil layers influences the quality of the shallow groundwater and the composition of the soil moisture in the rootzone. Climate and temperature also play a role in the rate of weathering and chemical processes.

Cropping patterns

Cropping patterns play an important role in the quality of drainage water in a number of respects. First, crops extract water from the rootzone resulting in an evapoconcentration of salts and other solutes in the soil solution. Where the solubility product of minerals is exceeded through evapoconcentration, minerals precipitate out. This changes the composition of the soil solution and thus influences the chemical quality of subsurface drainage waters. Second, crop residues add organic matter to the soil profile. Organic matter in the soil increases the adsorptive capacity for metals and other solutes. Furthermore, organic matter enhances the soil structure, which increases the water holding capacity of the soil. The organic matter also serves as a carbon source for soil microbes involved in transformations such as denitrification, sulphate reduction and methane production in submerged soils. Third, plants extract nutrients through their rooting system and some plants have the capacity to accumulate large amounts of certain salts and toxic elements. Fourth, as not all crops have the same salt tolerance the type of crop largely determines the maximum salt concentration in the rootzone and the amount of water needed to maintain a favourable salt balance in the rootzone. Last, incorporation of nitrogen fixing crops such as legumes can help to reduce nitrogen leaching. Legumes in symbiosis with nitrogen fixing bacteria are both users and producers of nitrogen. They can substitute chemical nitrogen fertilizer in the crop rotation. Deep-rooted perennial crops such as alfalfa can also help to prevent nitrogen leaching by absorbing large amounts of nitrogen (Blumenthal, *et al.*, 1999).

Use of agricultural inputs

Application of fertilizers, pesticides, soil and water amendments, and animal manures may influence the quality of drainage water to a great extent. The amounts and timing of application in relation to the growing stage of the crops, timing of irrigation, drainage practices and applied soil conservation measures largely define the influence of fertilizer, amendment and pesticide application on drainage water quality. Furthermore, the characteristics of the fertilizers themselves play a major role, also on the possible contamination of drainage water. Most nitrogen fertilizers are highly soluble and mobile in the soil, and nitrates readily enter drainage water through leaching processes. A portion of the nitrogen fertilizers made up of ammonium or anhydrous ammonia is initially adsorbed to the soil exchange complex but ammonium ions oxidize readily to nitrate. This is also true of urea containing nitrogen fertilizers that eventually oxidize to nitrates.

Conversely, phosphorus fertilizers are less mobile in the soil because they have very low solubility, and phosphates are adsorbed on positive sites in soil organic matter and clay minerals (Westcot, 1997). The main route for phosphorous to drainage water is through runoff as sediment-bound inorganic and organic phosphate. Runoff waters may contain residues of anhydrous ammonia injected into irrigation water. Ammonia is highly toxic to fish. Runoff waters may also contain sediment-bound organic nitrogen. Excessive levels of nitrogen and phosphate in discharge waters may result in the eutrophication of water bodies.

Water and soil amendments such as gypsum may contribute significantly to salinity. Although gypsum is sparingly soluble, calcium contributed from gypsum may exchange with adsorbed sodium, and sodium sulphate is a highly soluble mineral. Acidic amendments react with soil calcium carbonates but do not contribute to salinity because the solubility of calcium carbonate is very low. Where animal manures are mineralized, nitrates and salts are produced. Some animal manure (e.g. poultry manure) contains appreciable amounts of salts.

Irrigation and drainage management

Irrigation and drainage management are the main factors influencing the flow of water over and through the soil in arid zone croplands. As solute transport takes place mainly through soil water fluxes, irrigation and drainage management determine to a great extent the solute fluxes through the soil. In irrigation management, timing in relation to crop water requirements and to fertilizer and pesticide application is key to controlling the amount of soluble elements that will leach below the rootzone, from where they can be intercepted by subsurface drains. The timing of irrigation also affects capillary rise into the rootzone, which might cause an accumulation of salts in the rootzone but at the same time reduces the need for irrigation water. Of equal importance to the timing is the amount of irrigation water applied. Excess water, including infiltrated rainfall, leaches to deeper soil layers.

Drainage techniques and design

The choice of drainage technology and certain design choices influence the quality drainage effluent considerably. Subsurface drainage enhances the flow of water through the soil. Studies from the United States of America have shown that, in comparison to soils drained by means of surface drains, subsurface drainage reduces the amount of runoff and subsequently reduces the phosphorous contamination of surface water. At the same time, subsurface drainage enhances nitrogen leaching. Upon interception of leaching water, the nitrogen concentration of the final drainage effluent increases. In the same way, subsurface drainage intercepts other soluble elements present in the soil. This offers the possibility to control the concentration of harmful salts and toxic trace elements in the rootzone for optimal crop production but reduces the possibility for disposal and reuse of drainage water.

For water table and salinity control in irrigated lands, two types of drainage technologies are generally employed: subsurface tile drainage, and tubewell drainage. Subsurface tile drains are generally installed at a depth of less than 2 m below the soil surface. However, exceptionally, drains are installed at depths of up to 3 m. Tubewell drains vary between 6 and 10 m but might reach a depth of 100 m, e.g. deep tubewell drains in Pakistan. These differences in drain depth influence the quality of the effluent. The quality of drainage water from subsurface tile drains is influenced by the quality of irrigation water, the applied farm inputs and the quality of the shallow groundwater. In contrast, the quality of drainage effluent from tubewell drains is related mainly to the quality of the groundwater and to a lesser extent to the quality of irrigation water. Based on computer simulations in Pakistan, a comparison of tubewell and tile drainage showed that for Kairpur the salinity of drainage effluent was 11 750 and 2 100 ppm for tubewell and tile drainage, respectively, whereas for Panjnad Abbiana it was 22 918 and 610 ppm, respectively (Chandio and Chandio, 1995).

The depth of tile drain installation also influences the amount and quality of the generated drainage water. In the Aral Sea Basin, deep drains increase the flow that has to be evacuated and its salt content through mobilization of fossil groundwater through deep flow paths. For example, drainage water in the Karshi oblast collector is extremely saline (up to 8 g/litre). Expected mineralization levels of drainage water resulting from irrigation only are much less (case study Aral Sea Basin, Part II). Experiments in Australia with shallow and deep drains in a vineyard on a clay soil led to the same conclusion. Less drainage effluent was generated with shallow drains (0.7 m deep) than with deep drains (1.8 m deep). Moreover, the shallow drainage water had a lower salinity than the deep drainage water; together with reduced drainage volumes this resulted in a reduced salt load. As the deep drains removed much more salt than imported with the irrigation water (Table 8), this led to the conclusion that the salinity in the effluent of the deep drains was derived from deeper soil layers (Christen and Skehan, 2001).

Table 8
Salt applied in irrigation water and removed by drains

Treatment	Salt removed (kg/ha)	Salt removed/Salt applied*
Deep drains	607	11
Shallow drains	32	0.7

* Mean salt applied = 50 kg/ha
Source: Christen and Skehan, 2001.

Model simulations by Fio and Deverel (1991) confirm these observations. They showed that the base flow towards tile drains increased and the quality decreased with increasing installation depth under non-irrigated conditions. They also showed that this is a result of the path flows becoming deeper and longer with increased drain depth and spacing. On the other hand, during irrigation events the discharge of the shallow and deeper drains became similar as recharge due to deep percolation increased and the proportional contribution of deep groundwater to drain lateral flow decreased.

Characteristics of drainage water quality

Salts and major ions

The major cations and anions making up salinity are sodium, calcium, magnesium, potassium, bicarbonate, sulphate, chloride and nitrate. A lumped salinity parameter is frequently used such as *EC* in decisiemens per metre (dS/m) or *TDS* in milligrams per litre. The water quality of surface runoff typically deviates little from the composition of the irrigation water even if it flows over soils with visible salt crusts (Reeve *et al.*, 1955). On the other hand, deep percolation displaces salts accumulated in the soil profile from natural chemical weathering, blown in by salt dust, as well as evapoconcentrated salts derived from the applied irrigation water. Thus, the salt content of the collected subsurface drainage water mainly reflects the salinity characteristics of the soil solution, which in turn is influenced by soil parent material, salinity of the shallow groundwater and salts brought into the soil with irrigation water. In many places, the drainage water composition is further influenced by the mineral composition of deep groundwater which is intercepted by the drains.

Toxic trace elements

Trace elements are commonly present at low levels in nature. Many trace elements are essential micronutrients in very small quantities such as iron, manganese, molybdenum and zinc, but the range between deficiency and toxicity is narrow. Trace elements of concern in drainage water from irrigated lands include: arsenic, boron, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium, strontium, uranium, vanadium and zinc. The heavy metal trace elements are fixed strongly by soil materials and tend to be mobile only in the topmost soil layers. However, some of them form mobile metal-organic complexes in the presence of organic matter. Some trace elements (arsenic, selenium, molybdenum and uranium) are relatively immobile in the reduced form (precipitated or elemental) or are adsorbed while the oxidized and oxyanion species are mobile. For example, selenium is soluble in alkaline and well-oxidized soils.

Similar to the dissolved mineral salts, trace elements are evapoconcentrated in the presence of a growing crop as more or less pure water is lost into the atmosphere and the trace elements remain in the soil solution. Elevated concentrations of selenium, boron and molybdenum may be found in soils formed from Cretaceous shale and marine sedimentary rocks and their shallow groundwaters such as in the western side of the San Joaquin Valley. Selenium typically exists in a reduced form in geologic formations as seleniferous pyrite or organic forms of selenium. When the formations are exposed to the atmosphere, the reduced selenium oxidizes into soluble forms and may be subject to transport from irrigation and precipitation.

Naturally occurring arsenic is commonly found in volcanic glass in volcanic rocks of rhyolitic to intermediate composition (Hinkle and Polette, 1999). It may be adsorbed to and coprecipitated with metal oxides, adsorbed to clay minerals and associated with sulphide minerals and organic carbon (Welch *et al.*, 1988). In natural aquifers, arsenic is especially a problem in West Bengal, India, and Bangladesh but it reportedly also occurs in the United States of America, Hungary, Chile, China, Argentina, Ghana, Mexico, the Philippines, New Zealand and Mongolia.

To evaluate whether trace elements might potentially cause a problem in drainage water management, Westcot (1997) studied ranges and geometric mean values in soils for the priority pollutant trace elements. Comparing actual soil data with these levels would give an initial idea of the potential for trace element leaching.

Agropollutants

The two main agropollutants in agricultural drainage waters are nutrients and pesticides. As nutrients were covered above, this sections focuses on pesticides. Once a pesticide enters the soil, its fate is largely dependent on sorption and persistence (NRC, 1993). Sorption is mainly related to the organic carbon content of soils while persistence is evaluated in terms of half-life (the time taken for 50 percent of the chemical to be degraded or transformed). Pesticides with a low sorption coefficient and high water solubility are likely to be leached while pesticides with a long half-life could be persistent.

Pesticides vary widely in their behaviour. Pesticides that dissolve readily in water have a tendency to be leached into the groundwater and to be lost as surface runoff from irrigation and rainfall events. Pesticides with high vapour pressure are easily lost into the atmosphere during application. Pesticides that are strongly sorbed to soil particles are not readily leached but may be bound to sediments discharged from croplands. Pesticides may be chemically degraded through such processes as hydrolysis and photochemical degradation. Pesticides may be biologically degraded or transformed by soil microbes.

Therefore, many pesticides are mainly found in surface drainage water and not in subsurface drainage water as such, due to the filtering action of the soils. However, some pesticides have a tendency to be leached through soil profiles and accumulate in groundwater, such as organophosphates (e.g. DBCP and Atrazine).

Sediments

Sediment contamination is a main concern for surface drainage in hilly areas and in areas with high rainfall. Sediment production in arid zones occurs in improperly designed and managed surface irrigation systems, especially furrow irrigation. Sediments are a direct threat to living aquatic resources and the aquatic environment in general. They also increase the cost of drinking water treatment and maintenance of open surface drainage networks from sediment deposition like in Pakistan and China. In addition, phosphate, organic nitrogen and pesticides bound to sediment particles are a source of pollution. Sediment production can be reduced by minimum tillage practices (NRC, 1993) and limiting surface runoff through sound irrigation practices. Sediment settling ponds may be used to reduce the load of sediments in receiving waters. Polyacrylamides appear to serve as an excellent coagulant for sediments in farm drainage. Sediments are not normally found in subsurface drainage water. However, a drainage pipe filled with soil particles might cause sediment pollution in subsurface drainage water.

WATER QUALITY CONCERNS FOR WATER USES

Crop production

The total concentration of salts in drainage effluent is of major concern for irrigated agriculture. Salinity in the rootzone increases the osmotic pressure in the soil solution. This causes plants to exert more energy to take up soil water to meet their evapotranspiration requirement. At a certain salt concentration, plant roots will not be able to generate enough forces to extract water from the soil profile. Water stress will occur, resulting in yield reduction. The extent to which the plants are able to tolerate salinity in the soil moisture differs between crop species and varieties.

For the stability of the soil structure, the composition of the soil solution is an important factor. In the solid phase, soils have a net negative surface charge. The magnitude of the cation exchange capacity (*CEC*) depends on the amount and type of clay and the organic matter content. Cations such as calcium, magnesium, sodium, potassium and hydrogen are adsorbed on the exchanger sites. Normally, a large fraction of the adsorbed cations is divalent calcium and magnesium. Divalent cations adsorbed to clay minerals provide structure and stability. Where monovalent cations dominate the exchangeable cations (sodium in particular), the soil structure looses its stability and structural degradation occurs easily. As cations are mutually replaceable, the composition of the exchangeable cations is related to the proportion of cations present in the soil solution (Jurinak and Suarez, 1990). Therefore, where drainage water reuse for irrigation purposes is under consideration, not only the total salt concentration should be taken into account, but also the sodium to calcium and magnesium ratio, commonly expressed as the sodium adsorption ratio (*SAR*). High bicarbonate waters tend to precipitate out calcium carbonate. This may increase the *SAR* in the soil solution and increase the exchangeable sodium percentage (*ESP*) on the *CEC*.

The composition of the salts is also important for crop growth. Dominance of certain ions might cause an imbalance in ion uptake. This results in deficiencies of certain elements and depressed yields. The presence of high concentrations of sodium inhibits the uptake of calcium, causing nutritional disorders. Other ions can be toxic, causing characteristic injury symptoms as the ions accumulate in the plant. Toxic elements of major concern are chloride, sodium and boron (FAO, 1985b).

The extent to which crops suffer from salinity stress depends on several factors. Although yield reductions are defined as a function of the average salt concentration in the rootzone, interactions between soil, water and climatic conditions influence the relationship. Exceedingly high air temperatures may cause a reduced salt tolerance. Cultural practices also determine to a certain extent yield reduction resulting from salinity stress. Other plant characteristics (which differ between plant species, varieties of the same species and growth stages during which salinity stress occurs) determine their ability to cope with salinity stress.

The variations between crops in salt tolerance are attributable to the fact that certain crops can make the necessary osmotic adjustment to enable them to extract more water from saline soils. This adjustment involves two mechanisms: absorption of salts from the soil solution, and synthesis of organic solutes. Halophytes tend to absorb salts and impound them in the vacuoles, while organic solutes serve the function of osmotic adjustment in the cytoplasm. Normal plants tend to exclude sodium and chloride ions. For this reason, these plants need to rely more than halophytes on the synthesis of organic osmolytes. As a result, they are more salt sensitive than halophytes. Annex 1 presents data on crop tolerance to salinity and major ions.

Sensitivity to salts changes considerably during plant development. Most crops are sensitive to salinity during emergence and early development. Once established, most plants become increasingly tolerant during later stages of growth. There is general agreement that the earlier the plants are stressed, the greater the reduction in vegetative growth (Maas and Grattan, 1999).

Not all trace elements are toxic and small quantities of many are essential for plant growth (e.g. iron, manganese, molybdenum and zinc). However, excessive quantities might accumulate in plant tissues and cause growth reductions. Crop tolerance to trace element concentrations varies widely. When accumulated in plant tissue, certain trace elements are also toxic to animals and humans upon eating, e.g. selenium, arsenic and cadmium. As plants do not absorb most of the trace elements that are present in the soil, the trace elements accumulate in the soils.

In 1985, FAO published general guidelines for evaluating water quality for irrigated crop production (FAO, 1985b). These guidelines are general in nature and are based on numerous assumptions. Where the actual conditions differ substantially from those assumed, it might be necessary to prepare a modified set of guidelines. The case studies presented in Part II of this publication present several examples of guidelines developed for local conditions in the context of drainage water management. Pratt and Suarez (1990) provide a list of recommended maximum concentrations of trace elements for long-term protection of plants and animals.

Living aquatic resources, fisheries and aquaculture

Aquatic organisms have different requirements with respect to the chemical and physical characteristics of a water body. Dissolved oxygen, adequate nutrient levels, and the absence of toxic concentrations of hazardous elements are essential factors for sustaining aquatic life. Drainage water disposal can disturb the chemical and physical characteristics of the aquatic habitat.

In natural water, the levels of trace elements are normally very low. Elevated concentration levels have a negative impact and harmful effects on aquatic life. Some trace elements such as mercury and selenium are of particular concern because of their bioaccumulative nature, even at very low concentrations (Westcot, 1997). For example, in the United States of America the regulatory maximum contaminant level for selenium for aquatic biota in freshwaters is 2 ppb and for drinking-waters for humans, 50 ppb. The former is lower due to the bioaccumulation of selenium through the aquatic food chain.

Pesticides may also cause toxicity problems in aquatic organisms in surface waters. While pesticide use is currently highest in North America and Europe, it is expected to increase at a faster rate in developing countries in the near future. Many of the synthetic organic compounds are persistent and bioaccumulate. They magnify up the food chain and are often absorbed in body fat, where they can persist for a long time. In the case of fish tissues and fishery products, some of these compounds may also reach consumers. As fish are an important source of protein, it is essential to prevent and avoid accumulation of contaminants in fish or shellfish (Chapman, 1992).

All aquatic organisms including fish or other aquatic resources living in contaminated water bodies are being exposed daily to a multitude of synthetic chemical compounds that disrupt the development of the reproductive, immune, nervous and endocrine systems by mimicking hormones, blocking the action of hormones, or by other unknown interference with the endocrine system (Rutherford, 1997). Fish have different life stages: egg, larvae, fingerling and adult. Various pollutants may have different effects on their life cycle and on their functions and abilities (e.g. capacity to reproduce, nurse, feed and migrate).

The greatest threat to the sustainability of inland fishery resources is degradation of the environment. According to the GEO-I prepared by the UNEP, access to and pollution of freshwater are among the four key priority areas (FAO, 1999a). Various guidelines have been proposed for water important for fisheries or protecting aquatic environmental quality in general (EIFAC, 1964; British Colombia Ministry of Environment, Land and Parks, 1998; CCME, 1999a; and Chapman, 1992). Water quality guidelines established for temperate regions should not be applied without caution to other climate conditions as toxicity, persistence and accumulation rates might differ substantially (Biney *et al*., 1994).

Livestock production

Water for livestock watering should be of high quality to prevent livestock diseases, salt imbalance, or poisoning by toxic constituents. Many of the water quality variables for livestock are the same as for human drinking-water resources although the total permissible levels of total suspended solids and salinity may be higher (Chapman, 1992). Annex 2 presents water quality guidelines for livestock drinking-water quality. The guidelines consist of two parts. The first part consists of guidelines for the use of saline water for livestock and poultry. Unsafe levels of salinity and ions depend on the amount of water consumed each day, and on the type, weight, age and physical state of the animal (Soltanpour and Raley, 2001). The second part contains maximum recommended limits of both chemical and microbiological variables. These limits are based on animal health, quality of the products and taste.

Concerns for human health

The quality of water has a major influence on public health. Poor microbiological quality is likely to lead to outbreaks of infectious water-borne diseases and may cause serious epidemics. Chemical water quality is generally of lower importance. The impact of chemicals on human health tends to be of a chronic long-term nature, and there is time available to take remedial action. However, acute effects may be encountered where major pollution events occur or where levels of certain chemicals (e.g. arsenic) are high from natural sources (WHO, 2000).

Increases in salinity, related to drainage water disposal on a shared water resource, may threaten its use for domestic and drinking-water supply. Although the WHO (1993) has not formulated any guidelines based on *TDS*, high salt concentrations can cause taste problems. Concentrations of less than 1 000 mg/litre (1.56 dS/m) are normally acceptable to consumers. For the majority of the major ions, no health guidelines have been derived. Present guidelines are based on taste and other side-effects of individual ions, such as staining of laundry by iron, or the rotten egg smell of sulphidic water. Most of the toxic trace elements are included in the health criteria for guidelines for the quality of drinking-water as some of them are carcinogenic. For example, arsenic contamination of drinking-water supplies is of major concern in Bangladesh. Expected concentrations in natural waters are generally well below 1 mg/litre. Where concentrations are exceeded, expensive treatment processes are required to make the water acceptable for human consumption. Some well-known chemical pollutants that affect health include nitrate, arsenic, mercury and fluoride. In addition, there is an increasing number of synthetic organic compounds released into the environment whose effect on human health is poorly understood, but appears to be carcinogenic (WHO, 2000). Annex 3 presents WHO water quality guidelines.

Humans also use water resources for bathing and recreation. Such activities in contaminated waters pose a health risk due to: the possibility of ingesting small quantities; contact with the eye, nose and ear; and contact through open wounds. Health risks related to recreation are mainly related to pathogenic contamination. The potential risks from chemical contamination of recreational waters are usually small. Even repeated exposure is unlikely to result in ill effects at the concentrations of contamination found in waters and with the exposure patterns of recreational users. However, the aesthetic quality of recreational water is extremely important for the psychological wellbeing of users (WHO, 1998).

Chapter 5
Water conservation

NEED FOR WATER CONSERVATION MEASURES

Water conservation measures are the first-line option for the control and management of subsurface drainage water. Conservation measures involve reducing the amount of drainage water and they include: source reduction through sound irrigation water management; shallow water table management; groundwater management[1]; and land retirement. These measures affect other options such as the reuse and disposal of drainage water. In general, conservation measures consist of measures that aim to reduce the quantity of drainage effluent and measures that aim to reduce the mass emission of constituents into receiving water. Water quality impacts to water users (agriculture, fisheries, etc.) are reflected in terms of concentration but the control of drainage from irrigated lands is in terms of water volume and mass discharge of constituents.

The case studies from the United States of America, India, Pakistan and the Aral Sea Basin presented in Part II illustrate the need for source reduction. In the northern third of the San Joaquin Valley, limited drainage water disposal into the San Joaquin River is permitted in order to protect water quality for downstream water users. However, in the southern two-thirds of this valley there are no opportunities for any drainage into the river and the vadose zone has filled up with deep percolation. Subsurface drainage is practised with disposal into evaporation ponds. For the southern part of this valley, options such as deep-well injection, desalination, and water treatment to remove selenium are either generally too expensive for the farmers to bear the entire cost or technically not feasible. Therefore, source reduction plays a major role in dealing with problems caused by the shallow, saline groundwater in the San Joaquin Valley. In countries such as India and Pakistan, conservation measures are required as large tracts of land in need of drainage occur in inland basins without adequate disposal facilities. In Pakistan, under the umbrella of the Fourth Salinity Control and Reclamation Project, evaporation ponds have been constructed to relieve the disposal problem, though adverse impacts have been encountered in surrounding lands. To prevent serious environmental degradation and to sustain irrigated agriculture, parallel conservation measures need to be implemented.

When water is used conservatively, the concentration of salts and trace elements will rise in drainage waters but the mass emission rates will decrease because the volume of water discharged is smaller. As water is used conservatively and the leaching fraction is reduced, salts tend to accumulate in the rootzone. Under such conditions, the major concern is for rootzone salinity not to exceed crop salt tolerance. In the Nile Delta, Egypt, a situation of increasing concentration in soil and water salinity is anticipated as water is used more conservatively (Box 2). In smaller irrigated areas with relatively good natural drainage, water tables might drop sufficiently low, as a result of improved irrigation efficiency, to minimize capillary rise into the rootzone. Under

[1] High water table results from an imbalance in the water balance – water is being applied to the surface at a rate that exceeds the carrying capacity of the groundwater system, thereby raising groundwater levels. In groundwater management, groundwater pumping is increased in order to remove the excess groundwater and lower the water table (SJVDIP, 1999f). Application of this option needs substantial knowledge of groundwater systems and hydrology, which is beyond the scope of this publication.

these conditions, downward leaching of salts will result in an overall improved salt situation (Christen and Skehan, 2000).

Box 2: Need to increase water use efficiency as result of water scarcity – An example from Egypt

Estimated water balance for Egypt

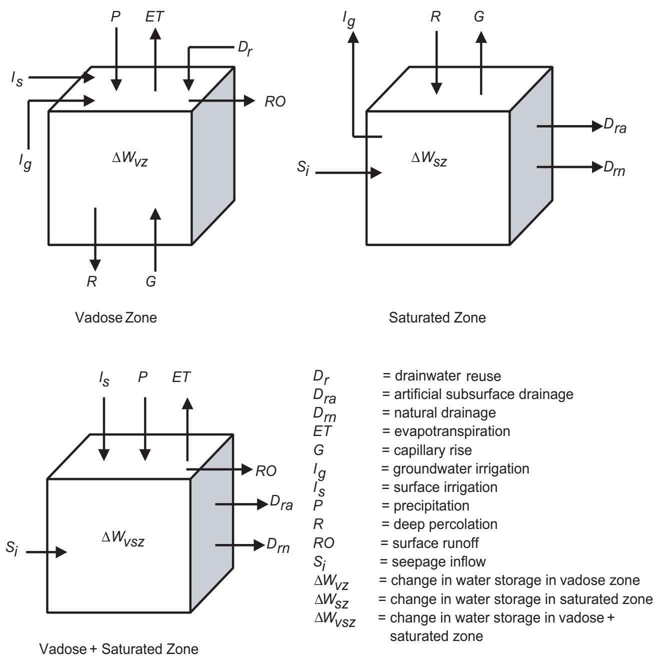
Water source	Annual availability in million m^3	Water use	Annual demand ('95) in million m^3
Nile water quota	55 500	Irrigation	51 500
Renewable GW	6 000	Municipal	2 900
Maximum reusable	7 000	Industrial	5 900
Total available	68 500	Total demand	60 300

The irrigation demand is projected to increase to 61 500 million m^3 in 2025. Without taking into account the increase in municipal and industrial demand, the total projected water demand for 2025 is 70 300 million m^3 per year. As the water resources are not projected to increase, irrigation efficiency needs to increase in order to sustain all valuable water uses. For this purpose, the Government of Egypt promotes the expansion of irrigation improvement projects. Reduction of irrigation losses reduces the drainage effluent generated and is likely to increase its salinity. This has considerable consequences for reuse and disposal (DRI, 1995; and EPIQ Water Policy Team, 1998).

Hydrologic balance

Drainage water management in general and water conservation measures in particular require a comprehensive knowledge and database of the hydrologic balance in irrigated agriculture. The control volume may be the vadose zone including the crop rootzone, the saturated zone for the groundwater basin, or a combination of vadose and saturated zones. The vadose and saturated zones may be viewed as the subsystem components and the combination of these two as the overall system or global perspective. Figure 15 depicts the hydrologic balance for these three control volumes. Water inputs and outputs vary among the subsystem components and the whole system. For example, pumped groundwater and drainage water reuse are vadose zone inputs but not considered in the combined vadose and groundwater system because they are internal flows that do not cut across the boundaries of the control volume.

Figure 15
Hydrologic balance in the vadose and saturated zones, and in a combination of vadose and saturated zones



The horizontal boundaries are normally easier to define because they often correspond to an irrigation district or a unit within an irrigation district. However, these types of boundaries are not always the most practical for defining the hydrological water balance as groundwater flow into and out of the district boundaries may be very difficult to estimate and measure. Defining the vertical boundaries is normally more difficult. The vertical components of the control volume include the crop rootzone, the vadose zone (the crop rootzone is often taken as part of the vadose zone) through which deep percolation passes and recharges the saturated or groundwater zone. Measurements of collected subsurface drainage are possible but deep percolation below the rootzone is extremely difficult to estimate and typically obtained as a closure term in the hydrologic balance in the vadose zone. The presence of a high groundwater table further complicates the situation as upward fluxes, i.e. capillary rise and water uptake from the shallow groundwater by roots, may also occur.

When viewing a series of interconnected irrigation and drainage districts as in a river basin, water use becomes quite complex. Although water may be used rather inefficiently upstream, the overall efficiency may be quite high for the entire river basin because of extensive reuse of water not consumed upslope or upstream, i.e. the irrigation return flow (Solomon and Davidoff, 1999). However, in terms of water quality, there is a progressive degradation because irrigation return flows pick up impurities (Chapter 4). Thus, it is necessary to be able to quantify the performance of irrigation and drainage systems not only for the design of alternative measures but also for the management and operation of irrigation and drainage systems.

Irrigation performance indicators

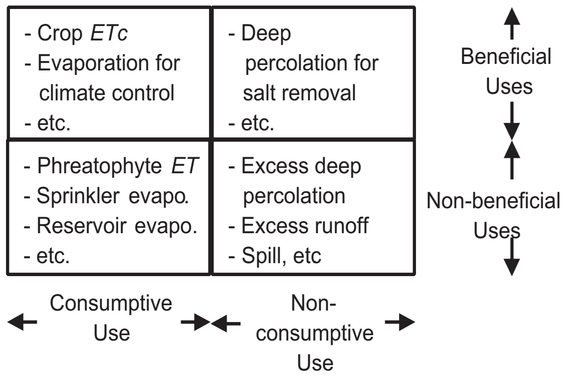
Numerous indicators, usually called efficiencies, have been developed to assess irrigation performance. The definitions of efficiencies as formulated by the ICID/ILRI (Bos and Nugteren, 1990) are widely used. According to the ICID/ILRI, the movement of water through an irrigation system involves three separate operations: conveyance, distribution, and field application. In congruence, the main efficiencies are defined for water use in each of these three operations. The conveyance efficiency (e_c) is the efficiency of the canal or conduit networks from the source to the offtakes of the distribution system. The e_c can be expressed as the volume of water delivered to the distribution system plus the non-irrigation deliveries from the conveyance system divided by the volume diverted or pumped from a source of water plus inflow of other sources. The distribution efficiency (e_d) is the efficiency of the water distribution (tertiary and quaternary) canals and conduits supplying water to individual fields. The e_d can be expressed as the volume of water furnished to the fields plus non-irrigation deliveries divided by the volume of water delivered to the distribution system. Finally, the water application efficiency (e_a) is the relationship between the quantity of water applied at the field inlet and the quantity of water needed for evapotranspiration by the crops to avoid undesirable water stress in the plants. Unlike the ASCE Task Committee on Describing Irrigation Efficiency and Uniformity, the ICID/ILRI do not make a further distinction between consumptive, beneficial and reasonable water use.

Source reduction aims to reduce the volume of drainage water by improving the irrigation performance. In addition to the efficiencies defined by the ICID/ILRI, this report uses a number of performance indicators defined by the ASCE Task Committee on Describing Irrigation Efficiency and Uniformity (Burt *et al.*, 1997) to assess the appropriateness of water use in an irrigation system and its contribution to the production of drainage water.

The ASCE Task Committee performance indicators are defined by the water entering and leaving the boundaries of a system. To judge the performance of an irrigation system, the ASCE Task Committee groups the fractions of water leaving the system boundaries in a specified time period into categories of use: consumptive versus non-consumptive use; beneficial versus non-beneficial use; and reasonable versus unreasonable use. This analysis requires an accurate description of the components of the hydrologic balance within defined boundaries of a control volume over a specific period of time. The time period taken is a cropping season rather than a single irrigation event.

Figure 16 shows that the division between consumptive use and non-consumptive use lies in the delineation between water that is considered irrecoverable and water that can be re-applied elsewhere, though perhaps degraded in quality. Consumptive uses include water that finds its way into the atmosphere through evaporation and transpiration, and water that leaves the boundaries in harvested plant tissues. Non-consumptive use is water that leaves the boundaries in the form of runoff, deep percolation and canal spills. The partitioning of applied water into beneficial and non-beneficial uses is a distinction between water consumed in order to achieve agronomic objectives and water that does not contribute to this objective. Beneficial use of water supports the production of crops and includes evapotranspiration, water needed for improving or maintaining soil productivity, seed bed preparation and seed germination, frost prevention, etc. Non-beneficial uses include deep percolation in excess of the leaching requirement, tailwater, evapotranspiration by weeds, canal seepage, spills, etc.

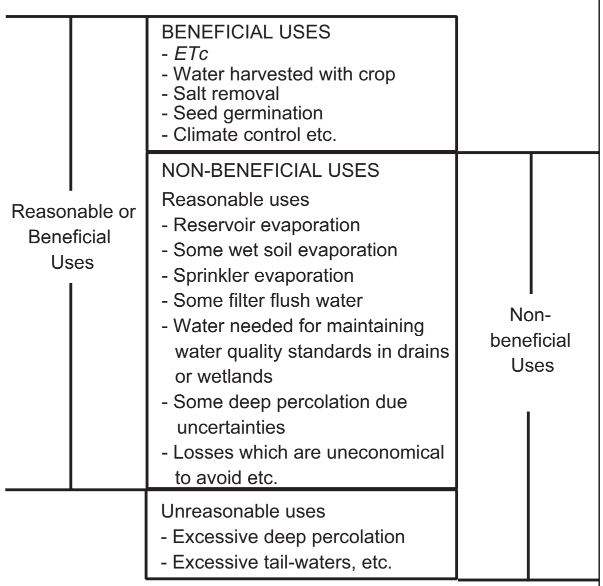
FIGURE 16
Consumptive versus non-consumptive and beneficial versus non-beneficial uses



Source: Clemmens and Burt, 1997.

Not all non-beneficial uses can be avoided at all times. Due to physical, economic or managerial constraints and various environmental requirements, some degree of non-beneficial use is reasonable. For example, reasonable but non-beneficial deep percolation can occur because of uncertainties that farmers face when deciding how much to irrigate to replenish the available soil moisture. Unreasonable uses include those that do not have any economical, practical or environmental justification. Figure 17 shows the division between beneficial and non-beneficial and reasonable and unreasonable uses.

FIGURE 17
Beneficial and non-beneficial and reasonable and unreasonable uses



Source: Clemmens and Burt, 1997.

Once defined, the water balance components can be used to make rational decisions about the appropriateness of the water use and whether they have a positive or negative impact on crop production, economy, regional hydrology, and on the amount of drainage water.

Three of the performance indicators proposed by the ASCE Task Committee are important in relation to the planning and design of conservation measures. The first indicator is the irrigation consumptive use coefficient (*ICUC*) which is expressed as follows:

$$ICUC = \frac{volume\ of\ irrigation\ water\ consumptively\ used}{volume\ of\ irrigation\ water\ applied - \Delta\, storage\ of\ irrigation\ water} * 100\% \qquad (6)$$

The *ICUC* deals with the fraction of water that is actually consumed in the system beneficially and non-beneficially. The denominator contains 'change in storage of irrigation water' or that part of the water applied that has not left the control volume and is thus usable during succeeding periods. The quantity (100-*ICUC*) represents the percentage of non-consumptive use of the applied water. Thus, the *ICUC* deals with the fraction of water that is irrecoverable and therefore cannot be re-applied elsewhere.

The second indicator, irrigation efficiency (*IE*) deals with water used beneficially for crop production. *IE* can be expressed as:

$$IE = \frac{volume\ of\ irrigation\ water\ beneficially\ used}{volume\ of\ irrigation\ water\ applied - \Delta\, storage\ of\ irrigation\ water} * 100\% \qquad (7)$$

The quantity (100-*IE*) represents the percentage of non-beneficial uses of applied water.

The ASCE Task Committee suggests a third performance indicator because not all water losses are avoidable. The irrigation sagacity (*IS*), first introduced by Solomon (Burt *et al*., 1997), is now defined by the ASCE Task Committee as:

$$IS = \frac{volume\ of\ irrigation\ water\ beneficially\ and\ /\ or\ reasonably\ used}{volume\ of\ irrigation\ water\ applied - \Delta\, storage\ of\ irrigation\ water} * 100\% \qquad (8)$$

The ASCE Task Committee suggests the use of *IS* to complement *IE*.

The numerical values of these three performance indicators provide appraisals on the overall effectiveness of the irrigation system and its management and the contribution towards drainage water production. However, the use of these performance indicators relies on quantification of the various water uses and fate pathways of water. There are many methods for determining the volume associated with each water use. Some are direct measurements (e.g. totalizing water meters), some are indirect measurements (e.g. *ET* estimated from weather data and crop coefficients) and some are obtained as the closure term from mass balance (e.g. deep percolation). Each method has errors associated with it that affects the accuracy of the performance indicator. Clemmens and Burt (1997) have shown that the confidence interval for the *ICUC* is about 7 percent. The accuracy of *IE* is in general less than the *ICUC*, as quantifying the beneficial water use is normally quite difficult. Therefore, they concluded that reporting more than two significant numbers for performance indicators is inappropriate. Furthermore, they introduced an approach using component variances to determine the relative importance of the accuracy of the variables that contribute to the estimate of the performance indicator. The component variances can be used to determine which measured volumes need closer attention. Improving the accuracy of the components with the highest variances will have the greatest impact on improving the accuracy of the performance measures.

SOURCE REDUCTION THROUGH SOUND IRRIGATION MANAGEMENT

Source reduction aims to reduce the drainage effluent through improving irrigation performance and thus improving *IE*. As certain losses are unavoidable, source reduction aims more precisely to increase *IS* by reducing non-beneficial, unreasonable uses (Box 3).

BOX 3: NON-BENEFICIAL UNREASONABLE USES

Non-beneficial Uses

- Any overirrigation due to non-uniformity;
- Any uncollected tailwater;
- Deep percolation in excess of that needed for salt removal;
- Evaporation from wet soil outside the cropped area of a field;
- Spray drift beyond the field boundaries;
- Evaporation associated with excessively frequent irrigations;
- Weed or phreatophyte evapotranspiration;
- System operational losses;
- Leakage from canals;
- Seepage and evaporation losses from canals and storage reservoirs;
- Regulatory spills to meet wastewater discharge requirements that are based on concentrations.

Unreasonable uses

- Non-beneficial uses that are also without any economic, practical or environmental justification. Unreasonable uses cannot be defined scientifically as they are judgmental and may be site and time specific. However, they should not be beyond engineering as engineering practice normally considers constraints, different objectives, economics, etc.

Source: Burt *et al.*, 1997

Reasonable losses

This section explores the various losses captured by subsurface field drains and open collector and main drains, and establishes estimates for reasonable losses.

Losses captured by subsurface field drains

The discharge of on-farm subsurface drainage in irrigated agriculture may be determined with the following water balance equation:

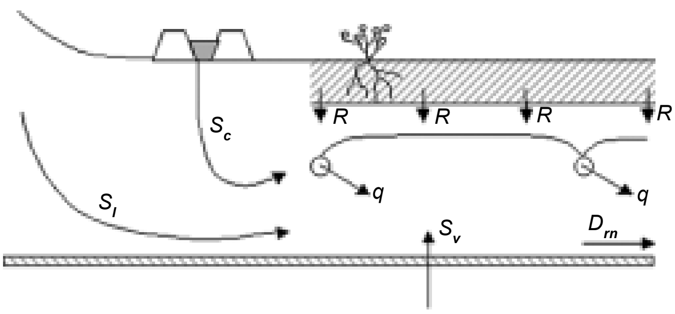
$$q = \frac{R + S_c + S_l + S_v - D_{rn}}{t} \tag{9}$$

where:

q = specific discharge (mm/d);
R = estimated deep percolation (mm);
S_c = seepage from canal (mm);
S_l = lateral seepage inflow to the area (mm);
S_v = vertical seepage inflow (mm);
D_{rn} = natural groundwater drainage from the area (mm); and
t = time period of measurement or calculation (d).

Figure 18 depicts these losses. In certain instances, the subsurface drainage system intercepts only a portion of the deep percolation.

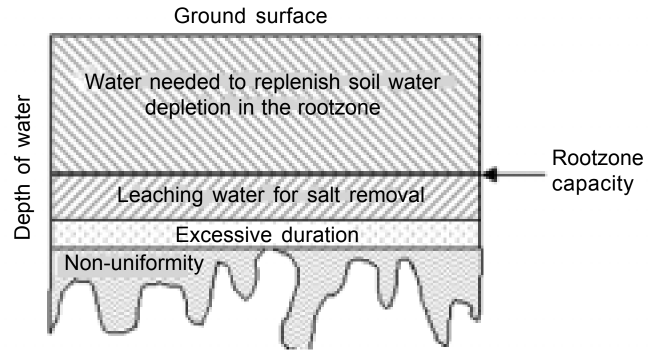
FIGURE 18
Losses captured by subsurface field drains



Deep percolation losses

Figure 19 shows that *R* consists of rootzone drainage from non-uniform water application, overirrigation from excessive duration of irrigation, water applied to leach salts, and rainfall. Unreasonable deep percolation losses are any percolation losses in excess of the inherent non-uniformity of the irrigation application system and salt leaching desired for the crop in question.

Figure 19
Deep percolation losses



Source: after Clemmens and Burt, 1997.

Table 9 shows that each irrigation method has a range of inherent distribution uniformity, e_a, and deep percolation. The data reported are based on well-designed and well-managed systems on appropriate soil types. In Table 9, e_a is defined as the ratio of the average amount of water stored in the rootzone to the average amount of water applied. Deep percolation, surface runoff, tailwater and evaporation losses make up the total losses. The e_a reported by Tanji and Hanson (1990) is based on data estimated from irrigated agriculture in California, the United States of America, where there is a state mandate to conserve water and where most irrigators receive some training.

Table 9
Estimated reasonable deep percolation losses as related to irrigation methods

Application method	Distribution uniformity (%)	Water application efficiency (%)		Estimated deep percolation (%)
		Tanji and Hanson, 1990	SJVDIP, 1999f	
Sprinkler				
-periodic move	70-80	65-80	70-80	15-25
-continuous move	70-90	75-85	80-90	10-15
-solid set	90-95	85-90	70-80	5-10
Drip/trickle	80-90	75-90	80-90	5-20
Surface				
-furrow	80-90	60-90	70-85	5-25
-border	70-85	65-80	70-85	10-20
-basin	90-95	75-90		5-20

Source: Tanji and Hanson, 1990; and SJVDIP, 1999e.

More recently, the Technical Committee on Source Reduction for the San Joaquin Valley Drainage Implementation Program (SJVDIP, 1999e) reported e_a values that differ somewhat from the 1990 values. The 1999 values were based on an analysis of nearly 1 000 irrigation system evaluations and they represent updated practical potential maximum irrigation efficiencies. In contrast, previously published FAO application efficiencies for irrigation application methods (FAO, 1980) and e_a values reported by the ICID/ILRI are substantially lower, representing a lower level of management and probably a less than optimal design. A properly designed system that is well managed can attain quite high efficiencies. On heavy soils, surface and drip irrigation can attain similar levels of efficiency.

Estimates for deep percolation have been made on the basis of the following assumptions: no surface runoff occurs under drip and sprinkler irrigation; during daytime sprinkler irrigation evaporation losses can be up to 10 percent and during night irrigation 5 percent; tailwater in furrow and border irrigation can be up to 10 percent and evaporation losses up to 5 percent; and no runoff occurs in basin irrigation and evaporation losses can be up to 5 percent.

Thus, reasonable deep percolation losses may vary with the irrigation application method and water management employed.

Canal seepage

Canal seepage varies with: the nature of the canal lining; hydraulic conductivity; the hydraulic gradient between the canal and the surrounding land; resistance layer at the canal perimeter; water depth; flow velocity; and sediment load. The canal seepage can be calculated using empirically developed formulae or solutions derived from analytical approaches (FAO, 1977). Canal seepage might also be estimated on the basis of Table 10.

TABLE 10
Seepage losses in percentage of the canal flow

Type of canal	Seepage losses (%)
Unlined canals	20-30
Lined canals	15-20
Unlined large laterals	15-20
Lined large laterals and unlined small laterals	10-15
Small lined laterals	10
Pipelines	0

Source: USBR, 1978.

Excessive seepage can occur due to poor canal maintenance. Any seepage in excess of the aforementioned figures needs to be regarded as unreasonable.

Seepage inflow

Seepage inflow from outside areas is discussed in the section on land retirement. Discussion of seepage inflow from deep aquifers requires substantial geohydrological knowledge, which is beyond the scope of this publication.

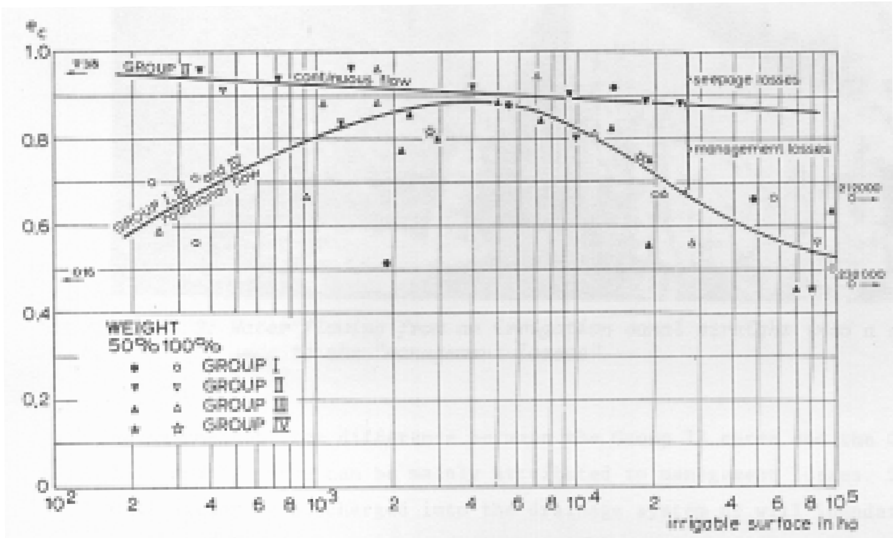
Losses captured by collector and main drains

The drainage discharge in collector and main drains depends on the discharge of field drains and additional inputs such as irrigation runoff from farmers' fields, and operational and canal spills. The ILRI in collaboration with the ICID analysed irrigation efficiencies for 91 irrigated areas on the basis of questionnaires submitted by 29 national committees of the ICID (Bos and Nugteren, 1990). For the interpretation of the data, the basic climate and socio-economic conditions were taken as primary variables. On the basis of these variables, the irrigated areas were divided into four groups. Group I includes all areas with severe rainfall deficit, and the farms are generally small with cereals as the main crop. For Group II, the main crop is rice and the rainfall deficit is less than for Group I. Group III has a shorter irrigation season than the first two groups and the economic development is more advanced. In addition to cereals, the most important crops are fodder crops, fruits and vegetables. For Group IV, irrigation is supplementary as this group has a cool, temperate climate.

The operational or management losses in the conveyance system are related to: the size of the irrigation scheme; and the level of irrigation management, communication systems and control structures, i.e. manual versus automatic control. Figure 20 shows that there is a sharp increase in operational losses in irrigation schemes of less than 100 ha and larger than 10 000 ha. Management losses can be as high as 50 percent.

The size of the tertiary or rotational units also has a significant influence on the operational losses. Bos and Nugteren (1990) estimate that optimum efficiency can be attained if the size of the rotational unit lies between 70 and 300 ha. Where the rotational units are smaller, safety margins above the actual amounts of water required are introduced, as the system cannot cope with temporary deficits. Larger rotational units require a long filling time in relation to the periods that the canals are empty, as the canals are relatively long and of large dimensions. This requires organizational measures to correct timing, which is often difficult.

FIGURE 20
Management losses in relation to the size of the irrigation scheme

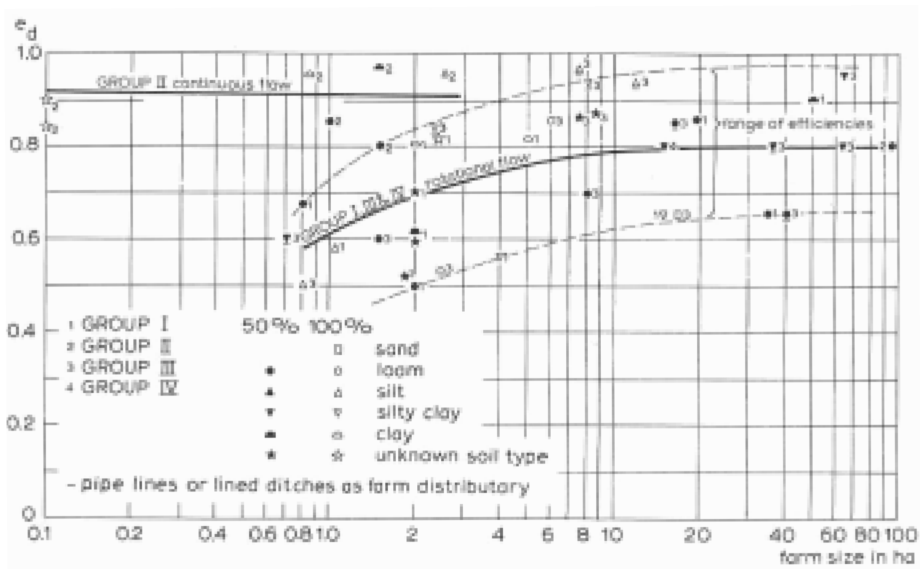


e_c = water conveyance efficiency
Source: Bos and Nugteren, 1990.

In Egypt, canal tail losses are estimated to account for 25-50 percent of the total water losses in irrigation (EPIQ Water Policy Team, 1998). For Egypt, it is expected that operational losses can be reduced significantly when measures such as automatic controls and night storage are introduced.

Other losses reaching collector and main drains are from the distribution system. In addition to the seepage losses from the tertiary and quaternary canals, the method of water distribution, farm size, soil type and duration of the delivery period affect the e_d. Figure 21 shows that the e_d is a function of farm size and soil type. Farm units of less than 10 ha served by rotational supply

FIGURE 21
Distribution losses in relation to farm size and soil type



Source: Bos and Nugteren, 1990.

have a lower efficiency than larger units. This is a result of the losses that occur at the beginning and end of each irrigation turn. Moreover, where farms are served by pipelines or are situated on less permeable soils, the e_d will be higher than average. Most of these losses do not occur if farms receive a continuous water supply. Consequently, in this case, the e_d is high irrespective of farm size.

When the delivery periods are increased, the e_d rises markedly. This is probably due to the losses that occur at the initial wetting of the canals.

Management options for on-farm source reduction

Improving on-farm irrigation management

Improving on-farm irrigation management involves optimizing irrigation scheduling. This means determining when to irrigate and how much water to apply. This can be done by using real-time weather data to obtain the reference crop evapotranspiration (*ETo*) and taking the product of *ETo* and crop coefficient to establish the depth of water to apply (SJVDIP, 1999e). Soil water balance methods are commonly used to determine timing and depth of future irrigations. The California Irrigation Management Information System (CIMIS) and FAO's CROPWAT (FAO, 1992a) are based on these principles. Basic assumptions in these methods are that healthy high-yielding crops are grown and that the soil moisture depletion between two irrigations equals the crop evapotranspiration. The latter assumption may not be valid where the plant roots are extracting water from shallow groundwater. Where such conditions exist, using CIMIS or CROPWAT will give an overestimation of the depletion and result in more water being applied than needed to replenish soil moisture. Field experiments have been implemented all over the world to evaluate the contribution of capillary rise from a shallow water table towards crop water requirements (e.g. Qureshi *et al*., 1997; DRI, 1997b; Minhas *et al*., 1988; and Rao *et al*., 1992). However, no simple calculation procedures have been developed yet. Ayars and Hutmacher (1994) propose a modified crop coefficient to incorporate groundwater contribution to crop water use. A manual calculation procedure is proposed in the section on shallow water table management. Rough estimates for capillary rise might be used, especially where drought resistant crops are grown or during periods when the crop is less sensitive to water stress. FAO (1986) has determined the yield response to water for a range of crops. Such information may be helpful in assessing the risks involved, in terms of yield losses, when rough estimates of capillary rise are used in optimizing irrigation scheduling.

Soil sampling and soil water sensing devices can provide valid estimates of soil moisture depletion. Provided these instruments are properly installed and calibrated, and the users adequately trained, irrigation scheduling can be based on the results.

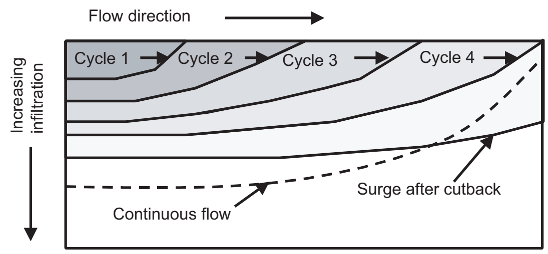
Improving water application uniformity and efficiency

Uniformity and e_a can be improved by upgrading the existing on-farm irrigation system or by converting to a technique with a potential for higher efficiency and uniformity (SJVDIP, 1999e). Surface irrigation by gravity flow is the most common irrigation technique as it does not involve the costs for the O&M of pressurized systems. For this reason, over the coming decades surface irrigation is likely to remain the predominant approach (FAO, 2000), although better uniformity and e_a can be obtained with sprinkler and drip irrigation.

There are several ways of improving the performance of furrow irrigation. The first method is to reduce the furrow length. This measure is most effective in reducing deep percolation below the rootzone for field lengths exceeding 300 m. The set time has to be reduced at the same time, and is equal to the difference in the initial advance time and the new advance time. Failure to do so will greatly increase the surface runoff and subsurface drainage. A second option is to apply cutback irrigation. Cutback irrigation means reducing the inflow rate of irrigated furrows after the completion of advance.

A third option is to use surge irrigation. Surge irrigation is intermittent application of water to an irrigation furrow (Yonts *et al.,* 1994). Initial infiltration rates in a dry furrow are high. As the water continues to run, the infiltration rate reduces to a constant rate. If water is shut off and allowed to infiltrate, surface soil particles consolidate and form a partial seal in the furrow, which substantially reduces the infiltration rate. When the water inflow into the furrow is re-introduced, more water moves down the furrow in the previously wetted area and less infiltration into the soil takes place. This process is repeated several times. As the previously wetted part of the furrow has a lower infiltration rate and the advance in this part is higher, the final result is a more uniform infiltration pattern (Figure 22).

FIGURE 22
Infiltration losses in furrow irrigation



Source: after Yonts *et al.*, 1994.

Last, better land grading and compaction of the furrow can improve the uniformity and efficiency of furrow irrigation (SJVDIP, 1999e).

Improvements in basin irrigation consist mainly of adjusting the size of the basins in accordance with the land slope, the soil type and the available stream size. FAO (1990b) gives guidelines on how to estimate optimal basin sizes. To obtain a uniformly wetted rootzone, the surface of the basin must be level and the irrigation water must be applied rapidly.

Drip and sprinkler irrigation systems have the potential to be highly efficient. However, where the systems are not properly designed, operated or maintained, the efficiency can be as low as in surface irrigation systems. Improving the uniformity and thus the efficiency in drip and sprinkler irrigation involves reducing the hydraulic losses. Losses can be minimized by selecting the proper length of the laterals and pipeline diameter and by applying appropriate pressure regulation throughout the system. After the system has been properly designed and installed, good O&M of the system is crucial. Phocaides (2001) provides guidelines on the design and O&M of pressurized irrigation techniques.

For additional details on on-farm irrigation management, the reader may refer to publications by Hoffman *et al.* (1990), and Skaggs and Van Schilfgaarde (1999).

Options for source reduction at scheme level

At a scheme level, numerous options can be applied to reduce the conveyance, distribution and operational losses. To reduce seepage and leakage, canal rehabilitation or upgrading might be required. To reduce the operational losses, improvements in the irrigation infrastructure and communication system could be implemented. To increase the distribution efficiency, tertiary and quaternary canals might need to be upgraded. Moreover, an increase in delivery period might help to increase the distribution efficiency. Furthermore, a number of policy options are available. In many countries, driven largely by financial constraints, the water users now manage the irrigation systems. It is hoped that handing over the systems to the water users will raise efficiency and profitability. FAO (1999b) has developed guidelines for the transfer of irrigation management services.

Impact of source reduction on long-term rootzone salinity

The main objective of source reduction in the context of drainage water management is to reduce the amount of drainage water. For the reduction of the amount of subsurface drainage this means that the amount of water percolating below the rootzone will be reduced through improving water application efficiency. In areas where salinization is a major concern, it is important to assess the feasibility and impact of source reduction on rootzone salinity.

The equilibrium rootzone salinity is a function of the salinity of the applied water that mixes with the soil solution and the fraction of water percolating from the soil solution (Annex 4). This can be expressed as:

$$EC_{f_r R} = EC_{SW} = \frac{EC_{IWi}}{LF_i} \tag{10}$$

where:

EC_{frR} = salinity of the percolation water which has been mixed with the soil solution (dS/m);

EC_{SW} = salinity of the soil water (dS/m);

EC_{IWi} = salinity of the infiltrated water that mixes with the soil solution (dS/m); and

LF_i = leaching fraction of infiltrated water that mixes with the soil solution (-).

Therefore, it is important to evaluate whether the average net amount of percolation water under proposed irrigation practices satisfies the minimum leaching requirements to avoid soil salinization. In this evaluation, rainfall should be considered as it might supply adequate leaching. The leaching fraction of the infiltrated water that mixes with the soil solution can be expressed as:

$$LF_i = \frac{f_r R^*}{f_i I_i + P_e} \tag{11}$$

where:

f_r = leaching efficiency coefficient of the percolation water (-);

f_i = leaching efficiency coefficient related to the incoming irrigation water that mixes with the soil solution (-);

I_i = irrigation water infiltrated, which is the total applied irrigation water minus the evaporation losses and surface runoff (mm);

P_e = effective precipitation (mm); and

R^* = net deep percolation (mm).

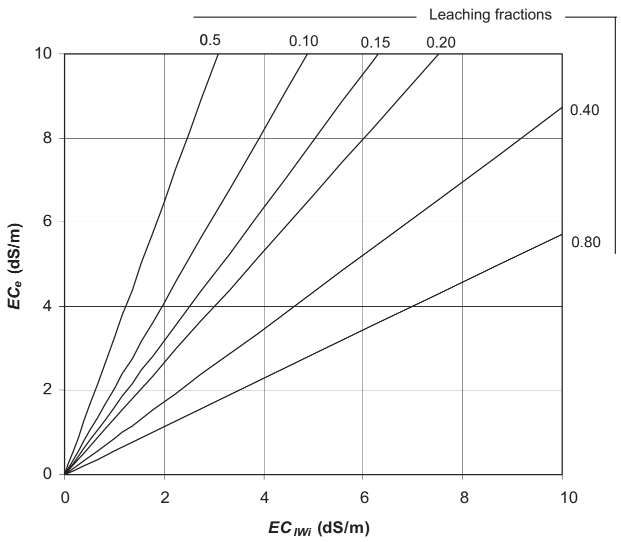
In this equation, it is assumed that over the irrigation seasons a shallow zone of water is created below the rootzone which has a salinity equivalent to the percolation water. Where this assumption is not correct or where deficit irrigation is actively practised under shallow groundwater conditions, capillary rise and deep percolation have to be entered as separate entities in the rootzone salt balance. In this case, the soil water salinity in the rootzone is a function of the salinity of the infiltrated water, capillary rise and percolation water.

It is often assumed that the salinity of the deep percolation water is equivalent to the average rootzone salinity. However, due to irrigation and rootwater extraction patterns, the salinity is lower in the upper portion of the rootzone due to higher leaching fractions (zone of salt leaching) and the salinity is higher in the bottom portions because of smaller leaching fractions (zone of salt accumulation). Under normal irrigation and root distribution, the typical extraction pattern for the rootzone is 40-30-20-10 percent water uptake from the upper to the lower quarter of the rootzone. Equation 10 can be used to calculate the rootzone salinity of five successive depths under this water uptake pattern to obtain finally the average salinity in the rootzone (Figure 2 in Annex 4).

Maintaining a favourable salt balance under source reduction

The relationship between EC_{IWi} and the average soil salinity of the saturated soil paste (EC_e) can be calculated for each LF_i and expressed as a concentration fraction (Table 1 in Annex 4). These concentration factors can be used to calculate the relationship between EC_e and EC_{IWi} (Figure 23). Where the salinity of the infiltrated water and the crop tolerance to salinity are known, the necessary *LF* to control soil salinity can be estimated from this figure. If the y-axis of the figure were the threshold salinity for the crop under consideration (EC_{ts}), then the diagonal lines would give a range of leaching requirements (*LR*) expressed as a *LF*.

FIGURE 23
Assessment of leaching fraction in relation to the salinity of the infiltrated water



Source: after FAO, 1985b.

Various researchers have developed other methods to calculate the *LR* to maintain a favourable salt balance in the rootzone. An empirical formula developed by Rhoades (1974) and Rhoades and Merrill (1976) is:

$$LR = \frac{EC_{IW}}{5 * EC_{ts} - EC_{IW}} \qquad (12)$$

where EC_{ts} is the threshold salinity for a crop above which the yield begins to decline (Annex 1) and EC_{IW} is the salinity of the infiltrated water. The value 5 was obtained empirically (FAO, 1985b).

The salt equilibrium equation (Equation 10) can be used to assess the feasibility and impact of source reduction on the rootzone salinity by calculating the *LR* expressed as *LF*, and to compare the results with the expected percolation under improved irrigation practices. The amount of percolation water should cover the *LR*. A safety margin is advisable because irrigation uniformity is never complete in the field (FAO, 1980). Depending on the distribution uniformity of the application method, the actual *LR* should be 1.1-1.3 times higher than the calculated *LR*. For example, for drip, sprinkler (solid set) and basin irrigation, smaller safety margins might be adopted while for border irrigation a larger margin might be more appropriate (Table 9).

Below an illustrative calculation from Pakistan is given. The example shows the impact of source reduction on the rootzone salinity.

Calculation example impact of source reduction on salinity of rootzone

General information

The drainage pilot study area is situated in the south east of the Punjab, Pakistan. The area has been suffering from waterlogging and salinity problems for a long time and therefore was selected as a priority area in urgent need of drainage. The main causes of waterlogging are overirrigation and seepage inflow from surrounding irrigated areas and canals. The estimated average seepage inflow is 0.5 mm/d with a salinity of 5 dS/m. In 1998, a subsurface drainage system was installed on a pilot area of 110 ha. The design discharge of the system is 1.5 mm/d and the design water table depth is 1.4 m. The soils are predominantly silty (θ_{fc} 0.36) with an estimated leaching efficiency coefficient (f_i) of 0.9. The climate is semi-arid with an annual potential crop evapotranspiration of 1 303 mm and an average annual effective rainfall of 197 mm. The main crops are wheat in winter (December to April) and cotton in summer (June to October/November). Both crops are irrigated throughout the year, as rainfall is insufficient to meet crop water requirements (Table 11). Irrigation water is supplied through open canals and has a salinity of 1 dS/m. Wheat is irrigated through basin irrigation and cotton is irrigated with furrow basins. The major application losses occur as a result of deep percolation due to poor field levelling (non-uniformity) and as a result of uncertainties in relation to rainfall and water distribution. The annual average e_a is 64 percent $((ET_{crop}-P_e)/I)$, including 5 percent evaporation losses. The depth of the rootzone is assumed to be 1 m and the water uptake pattern is 40, 30, 20, 10 percent from the first to the fourth quarter of the rootzone, respectively. It is assumed that no capillary rise occurs.

Long-term rootzone salinity

Long-term salinity in the rootzone under these conditions is calculated using Equations 1-11 presented in Annex 4 for the four-layer concept. The first step is to calculate EC_{IWi} using Equation 10:

$$EC_{IWi} = f_i I_i / (f_i I_i + P_e) * EC_I = (0.9 * 0.95 * 1718) / (0.9 * 0.95 * 1718 + 197) * 1 = 0.88 \text{ dS/m}$$

The second step is to calculate the LF_i values for the infiltrated water mixing with the soil solution for the successive quarters (Figure 2 in Annex 4):

$$LF_{i1} = (IW_i - 0.4 * ET_{crop}) / IW_i = (0.9 * 0.95 * 1\,718 + 197 - 0.4 * 1\,303) / (0.9 * 0.95 * 1\,718 + 197) = 0.69$$

$$LF_{i2} = (IW_i - (0.4 + 0.3) * ET_{crop}) / IW_i = (1\,666 - 0.7 * 1\,303) / 1\,666 = 0.45$$

$$LF_{i3} = (IW_i - (0.4 + 0.3 + 0.2) * ET_{crop}) / IW_i = (1\,666 - 0.9 * 1303) / 1\,666 = 0.30$$

$$LF_{i4} = (IW_i - (0.4 + 0.3 + 0.2 + 0.1) * ET_{crop}) / IW_i = (1\,666 - 1.0 * 1\,303) / 1\,666 = 0.22$$

TABLE 11
Agroclimatic data for the drainage pilot study area

Period	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Year
Crop	wheat						cotton						
ET (mm)	43	55	101	110	76	61	145	154	219	195	114	30	1303
P_e (mm)	5	15	15	9	5	15	80	35	10	0	3	5	197
I (mm)	222	122	122	122	0	144	144	144	288	144	144	122	1718

The third step is to calculate the EC_e values with Equation 7 (Annex 4). Considering that for most soils $EC_e \approx 0.5\ EC_{sw}$:

$EC_{e1} = 0.5\ EC_{sw1} = 0.5 * 0.88 / 0.69 = 0.64$ dS/m

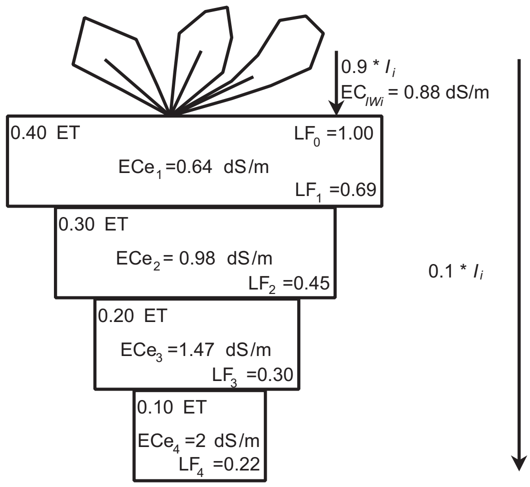
$EC_{e2} = 0.5\ EC_{sw2} = 0.5 * 0.88 / 0.45 = 0.98$ dS/m

$EC_{e3} = 0.5\ EC_{sw3} = 0.5 * 0.88 / 0.30 = 1.47$ dS/m

$EC_{e4} = 0.5\ EC_{sw4} = 0.5 * 0.88 / 0.22 = 2.00$ dS/m

The average EC_e of the rootzone, including the first value of the infiltrated water, is 1.11 dS/m. Figure 24 presents the results of these calculations.

FIGURE 24
Calculation of average rootzone salinity for the drainage pilot study area



Source reduction and the impact on rootzone salinity

Disposal of drainage water in southern Punjab is a major problem. Only very limited volumes of drainage water can be disposed into main irrigation canals and rivers. Evaporation ponds have been constructed to relieve the disposal problem but adverse environmental effects are observed. To sustain irrigated agriculture and to prevent serious environmental degradation, parallel conservation measures have to be implemented. As the calculated rootzone salinity of 1.11 dS/m is far below the threshold salinity value of wheat and cotton, source reduction at field level should be considered. Source reduction might be attained through a combination of measures including: precision land levelling; the shaping of basin and basin furrows; improved irrigation water distribution; and the introduction of water fees.

The threshold EC_e value for wheat is 6 dS/m and for cotton 7.7 dS/m (Annex 1). To assess the minimum leaching requirement, the threshold rootzone salinity of the most sensitive crop in the crop rotation will be used. Using Figure 23, theoretically, the leaching fraction of the percolation water mixing with the soil solution could be less than 5 percent. If the LF_i value at the bottom of the rootzone is assumed to be 0.05, the total irrigation water applied can be calculated as:

$LF_i = (f_i I_i + P_e - ET_{crop}) / (f_i I_i + P_e)$

$0.05 = (0.95 * 0.9 * I + 197 - 1\ 303)/(0.95 * 0.9 * I + 197)$

$0.05 * (0.855 * I + 197) = (0.855 * I - 1\ 106)$

$I = 1\ 374$ mm

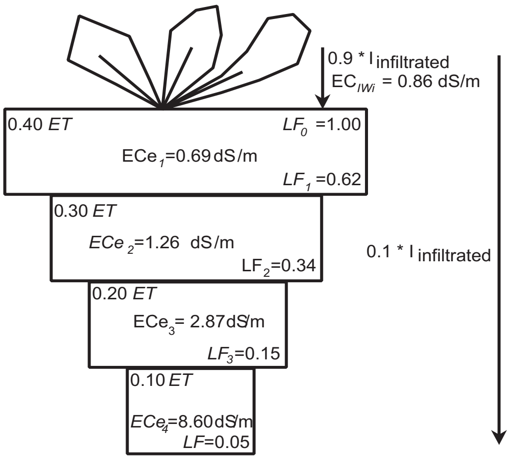
In this case, the e_a is 80 percent, which is within the range of reasonable deep percolation losses (Table 9). Higher efficiencies cannot be achieved as farmers will need to cope with uncertainties in rainfall and to a certain extent in water supply. Second, some losses always occur as a result of non-uniformity and farmers' inability to apply exact amounts of water. Equation 10 (Annex 4) can be used again to calculate the salinity of the infiltrated water mixed with the soil water:

$EC_{IWi} = (0.9 * 0.95 * 1\ 374) / (0.9 * 0.95 * 1\ 374 + 197) * 1 = 0.86$ dS/m

The *LF* values for the infiltrated water mixing with the soil solution and the EC_e values for the successive quarters are calculated the same way as for Figure 24 and result in an average rootzone salinity of 2.77 dS/m (Figure 25) against the original value of 1.11 dS/m. Indeed, the salinity increased, but it is still acceptable for the common crops grown in the area.

The above calculations were based on dividing the rootzone into four equal parts (quadrants). However, this model can be extended into n-number of layers provided that the rootwater extraction pattern is known, e.g. in 15-cm depth increments for 90-cm rooting depth.

FIGURE 25
Calculation average rootzone salinity under source reduction in the drainage pilot study area



Impact of source reduction on salt storage within the cropping season

In previous sections, long-term steady-state conditions were assumed to prevail. However, salinity levels during a cropping season are not stable and change throughout the season. To study the impact of irrigation and drainage measures on crop performance, it is important to know the changes in rootzone salinity during a cropping season over multiple time periods. These periods may range from a day to a monthly period. The mass of salts at the start of a period and at the end of a period normally differs and can be expressed as:

$$S_{start} = S_{end} - \Delta S \quad (13)$$

where:

S_{start} = quantity of salts in the rootzone at the start of the period (ECmm[1]);
S_{end} = quantity of salts in the rootzone at the end of the period (ECmm); and
ΔS = change in salt storage in the rootzone (ECmm).

Equations 16 to 24 in Annex 4 provide a full description of the derivation of the salt storage equation. The salt storage equation is defined as:

$$\Delta S = \frac{IW\,EC_{IW} - \dfrac{R^* S_{start}}{W_{fc}}}{1 + \dfrac{R^*}{2W_{fc}}} \quad (14)$$

where:

IW = infiltrated water (mm); and
W_{fc} = moisture content at field capacity (mm).

[1] The quantity ECmm requires some explanation. The parameter *S* is the mass of salts obtained from the product of salt concentration and water volume per area. For the sake of convenience, Van Hoorn and Van Alphen (1994) chose to use *EC* instead of *TDS* in grams per litre. The unit millimetre equals litre per square metre. Thus, the parameter *S* corresponds with the amount of salt in grams per square metre.

Equation 14 can be used to calculate changes in soil salinity within a cropping season. The salt storage equation can also be applied to the four-layer concept. Using the data from the previous example, the following calculation example assesses the impact of source reduction on the rootzone salinity on a monthly basis.

Calculation example of impact of source reduction on salt balance of the rootzone

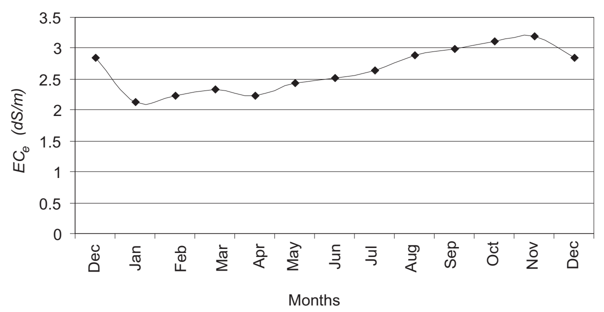
To explore the changes in salinity over the growing season as a result of source reduction, the salt storage equation (Equation 14) applied to the four-layer concept as explained in Annex 4 is used. For each layer the change in salt storage is calculated using Equation 22 as presented in Annex 4 for the first layer and Equation 23 for the consecutive three layers. The average EC_e is based on the quantity of salts stored in the rootzone at the end of the calculation periods. Equation 19 presented in Annex 4 is used to convert S to EC_e. Table 12 presents the results for the wheat-growing season.

TABLE 12
Salt balance in the root zone for the drainage pilot study area during wheat season

Crop	Wheat				
Period	**Dec**	**Jan**	**Feb**	**Mar**	**Apr**
ET_{crop} (mm)	43	55	101	110	76
P_e (mm)	5	15	15	9	5
I_i (mm)	163.4	91.2	91.2	91.2	0
EC_{IWi} (dS/m)	0.97	0.85	0.85	0.90	0.00
ΔW_1 (mm)	0	0	0	0	-25.4
R_1 (mm)	135	75	57	47	0
LF_1 (-)	0.89	0.77	0.58	0.52	0.00
$S_{start\ 1}$ (ECmm)	131	103	100	115	132
ΔS_1 (ECmm)	-28	-3	14	18	0
$S_{end\ 1}$ (ECmm)	103	100	115	132	132
$R_1\ EC_{R1}$ (ECmm)	175	85	68	65	0
ΔW_2 (mm)	0	0	0	0	-22.8
R_2 (mm)	122	59	26	14	0
LF_2 (-)	0.90	0.78	0.47	0.30	0.00
$S_{start\ 2}$ (ECmm)	317	165	148	169	204
ΔS_2 (ECmm)	-151	-17	21	35	0
$S_{end\ 2}$ (ECmm)	165	148	169	204	204
$R_2\ EC_{R2}$ (ECmm)	326	102	46	29	0
ΔW_3 (mm)	0	0	0	-8	-23
R_3 (mm)	101	48	6	0	0
LF_3 (-)	0.83	0.81	0.23	0.00	0.00
$S_{start\ 3}$ (ECmm)	840	445	340	362	391
ΔS_3 (ECmm)	-395	-106	22	29	0
$S_{end\ 3}$ (ECmm)	445	340	362	391	391
$R_3\ EC_{R3}$ (ECmm)	721	207	24	0	0
ΔW_4 (mm)	0	0	-4	-15	-23
R_4 (mm)	26	42	0	0	0
LF_4 (-)	0.26	0.88	0.00	0.00	0.00
$S_{start\ 4}$ (ECmm)	1 506	1 758	1 260	1 284	1 284
ΔS_4 (ECmm)	252	-498	24	0	0
$S_{end\ 4}$ (ECmm)	1 758	1 260	1 284	1 284	1 284
$R_4\ EC_{R4}$ (ECmm)	468	706	0	0	0
$\overline{EC_e}$ (dS/m)	2.84	2.14	2.23	2.33	2.24

Figure 26 shows the changes in the average rootzone salinity over an entire year. The salinity increases during the summer months and reaches its peak towards the end of the cotton growing season. Before planting wheat in November-December, many farmers apply a rauni (pre-irrigation) to leach any accumulated salts.

FIGURE 26
Change in the average rootzone salinity over the year under source reduction



Impact of source reduction on salinity of drainage water

To estimate the impact of source reduction on the extent of reuse and disposal of drainage water, it is necessary to consider not only the water quantity but also the changes in water quality. Generally, the salinity of the drainage water increases as the drainage discharge decreases. Where all the deep percolation is intercepted by the subsurface drainage system, the amount of generated drainage water is:

$$D_{ra} = fr R^* + (1\text{-}fr) R^* + (S_i - D_{rn}) \tag{15}$$

The salt load of the subsurface drainage water is the salinity of the intercepted water multiplied by the depth of drainage water plus the salt load of the seepage inflow minus the salt load of the natural drainage. If the natural drainage can be ignored the salt load of the subsurface drainage salinity can be calculated as follows:

$$D_{ra} EC_{Dra} = fr R^* EC_{frR} + (1\text{-}fr) R^* EC_I + S_i EC_{S_i} \tag{16}$$

where:

EC_{Dra} = salinity of the subsurface drainage water (dS/m);
EC_{frR} = salinity of the percolation water that mixed with soil solution (dS/m); and
EC_{Si} = salinity of the seepage inflow intercepted by the subsurface drains (dS/m).
EC_I = salinity of the irrigation water (dS/m)

The salinity of the subsurface drainage water (EC_{Dra}) is the salt load as calculated with Equation 16 divided by the depth of drainage water.

In the California studies, there appears to be some correlation between salinity, boron and selenium in terms of the waste discharge load, i.e. changes in flow result in similar changes in loads of salts, selenium and boron. This load-flow relationship exists because the shallow groundwater contains excessive levels of salinity, selenium and boron (Ayars and Tanji, 1999). Where this relationship exists in an area, regulating the salinity load would also regulate boron and selenium loads in drainage discharge.

The salinity of the subsurface drainage water is diluted in the main drainage system by surface runoff and canal spills. These losses will be drastically reduced under irrigation improvement projects. For example, in Egypt it is expected that through the introduction of night storage reservoirs, the canal tailwater losses will be reduced to almost zero (EPIQ Water Policy Team, 1998).

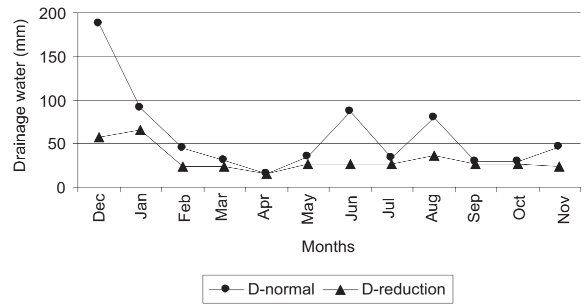
Calculation example of source reduction and the impact on drainage water generation and salinity

Source reduction influences not only the rootzone salinity but also the amount of drainage water generated and the salinity of the drainage water. This is of special interest when seeking options for reuse and disposal. Under normal conditions and source reduction, the amount of generated drainage water in the drainage pilot study area in Pakistan can be calculated using Equation 15:

D_{ra}-normal = 363 + 163 + 182.5 = 708.5 mm
D_{ra}-reduction = 69 + 131 + 182.5 = 382.5 mm

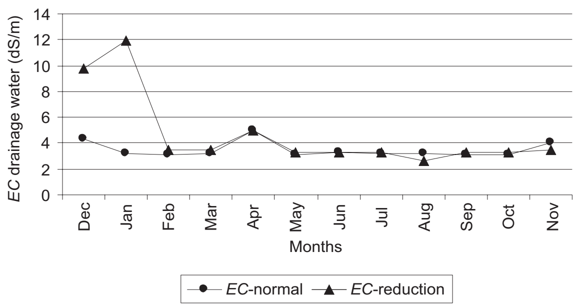
Thus, when source reduction is applied, 46 percent less drainage water is produced. A major reduction is obtained in the first months of the winter season and to a lesser extent during the summer season (Figure 27).

FIGURE 27
Depth of drainage water generated



The salinity of the drainage water increases mainly during the first two months of the winter season when source reduction is applied in comparison to the normal irrigation practices (Figure 28). The increased rootzone salinity that develops over the summer season as a result of reduced leaching and the subsequent leaching at the start of the winter season are the main reasons for this increase in salinity of the drainage water.

FIGURE 28
Salinity of the generated drainage water



The total salt load is normally of interest for disposal options. The salt load can be expressed as the depth of water multiplied by the salinity of the drainage water (ECmm). The annual salt load is calculated using Equation 16:

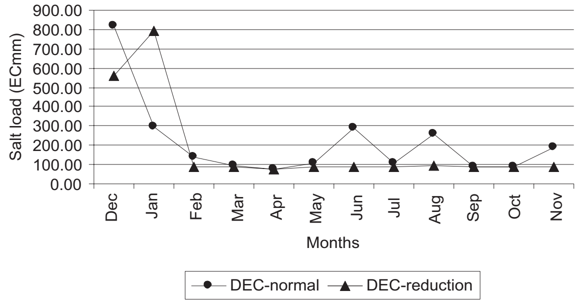
$D_{ra}EC_{Dra}$-normal = 363 * 2 * 2.0 + 163 * 1 + 182.5 * 5 = 2 528 ECmm
$D_{ra}EC_{Dra}$-reduction = 69 * 2 * 8.60 + 131 * 1 + 182.5 * 5 = 2 230 ECmm

This means that the annual salt load reduction achieved under source reduction is 11.8 percent. If the *TDS* (g/litre) is approximately 0.64 *EC* (dS/m), the annual reduction is salt load is 1.9 tonnes per ha. Figure 29 shows the distribution of salt load over the year.

SHALLOW WATER TABLE MANAGEMENT

In the past, drainage systems were typically designed to remove deep percolation from irrigated land plus any seepage inflow. In saline groundwater areas the depth and spacing was also determined to minimize potential capillary rise of saline groundwater into the rootzone. However, the interaction between crop water use from shallow groundwater and irrigation management has been used to demonstrate the potential impact on irrigation design and the active management of drainage systems. However, there has been little research on the active management of drainage systems in arid and semi-arid areas (Ayars, 1996). The concept of controlled subsurface drainage was first developed for humid climates.

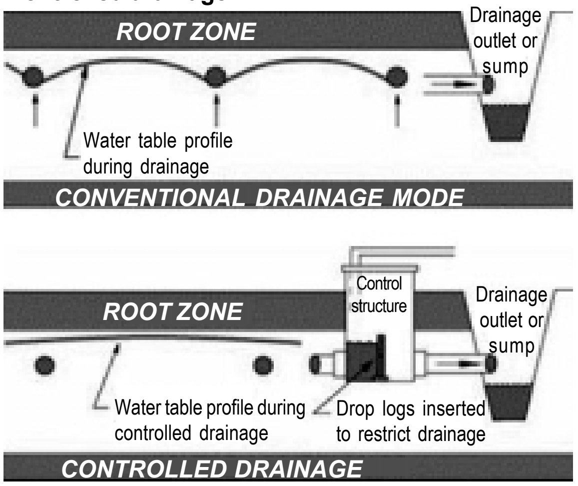
FIGURE 29
Salt load in the generated drainage water



Controlled subsurface drainage

The concept of controlled subsurface drainage can be applied as a means to reduce the quantity of drainage effluent. Figure 30 shows how a control structure at the drainage outlet or a weir placed in the open collector drain allows the water table to be artificially set at any level between the ground surface and the pipe drainage level, so promoting rootwater extraction. The size of the areas where the water table is controlled by one structure depends on the topography. During system operation, it is important that the water table be maintained at a relatively uniform depth. A detailed topographical map is essential for dividing an area into zones of control for the water management system (Fouss *et al*., 1999). Butterfly valves could be installed to restrict the flow from individual lateral lines, and manholes with weir structures could be installed along the collector (Ayars, 1996). In situations where the drain laterals run parallel to the surface slope, in-field controls might be necessary to maintain uniform groundwater depths (Christen and Ayars, 2001).

FIGURE 30
Controlled drainage



Source: Zucker and Brown, 1998.

Raising the outlet or closing the valves at predetermined times maintains the water table at a shallow depth. Withholding irrigation applications at the same time induces capillary rise into the rootzone. In this way, plants meet part of their evapotranspiration needs directly from soil water, replacing irrigation water with shallow groundwater that would otherwise have been evacuated by the drainage system. Without irrigation applications, the water table gradually drops in areas with no seepage inflow from outside, while a more or less constant water table can be maintained in areas with high seepage inflow.

Where shallow water table management through controlled drainage is practised, the drainage effluent and salt load discharged is reduced. Controlled drainage also helps to reduce the loss of nutrients and other pollutants in subsurface drainage water (Skaggs, 1999; and Zucker and Brown, 1998). Maintaining a high groundwater table significantly decreases the nitrate concentration in the drainage water. This decrease is a result of a load reduction in drainage discharge and an increase in denitrification.

Considerations in shallow water table management

> **BOX 4: CONTRIBUTION OF CAPILLARY RISE IN INDIA**
>
> In India, in a sandy loam soil with a water table at 1.7 m depth and a salinity of 8.7 dS/m, capillary rise contributed up to 50 percent of the crop water requirement. Similarly, at another site a shallow water table at 1.0 m depth with salinity in the range of 3.5-5.5 dS/m facilitated the achievement of the potential crop yields whilst reducing the irrigation application by 50 percent. In both cases, the accumulated salts were leached in the subsequent monsoon season (case study on India, Part II).

In arid to semi-arid areas the main purpose of drainage is the control of waterlogging and salinity. Maintaining the water table at a shallow depth and thus inducing capillary rise into the rootzone seems counterproductive to attaining this objective. However, projects in Pakistan and India have shown that the salts, which accumulated in the rootzone when shallow groundwater was used for evapotranspiration, were easily leached before the next cropping season. In monsoon-type climates, the accumulated salts are leached in the subsequent monsoon season (Box 4). This experience shows that shallow water table management is an important mechanism for reducing the drainage effluent and at the same time for saving water for other beneficial uses.

The extent to which shallow water table management can be used to reduce the drainage effluent discharge depends on:

- capillary rise into the rootzone, which is related to the water table depth, soil type and water table recharge;
- salt accumulation in relation to the salt tolerance of the crops; and
- potential to maintain a favourable salt balance.

Capillary rise

Capillary flow depends on: soil type; soil moisture depletion in the rootzone; depth to the water table; and recharge. Evapotranspiration depletes the soil moisture content in the rootzone. Where no recharge through irrigation or rainfall takes place, a difference in potential induces capillary rise from the groundwater. In the unsaturated rootzone, the matrix head, which is caused by the interaction between the soil matrix and water, is negative. At the water table, atmospheric pressure exists and therefore the matrix head is zero. The water moves from locations with a higher potential to locations with lower potentials. Capillary rise from the groundwater or underlying soil layers to the rootzone takes place under the influence of this head difference.

Darcy's Law (Annex 5) can be applied to calculate the capillary flow. However, the unsaturated hydraulic conductivity, $K(\theta)$, needs to be known. A difficulty is that $K(\theta)$ is a function of the moisture content and the moisture content is a function of the pressure head.

Although water movement in the unsaturated zone is in reality unsteady, calculations can be simplified by assuming steady-state flow during a certain period of time. The steady-state flow equation can be written as:

$$z_2 = z_1 + \left(\frac{h_2 - h_1}{-1 - \dfrac{q}{K\left(\bar{\theta}\right)}} \right) \qquad (17)$$

where:

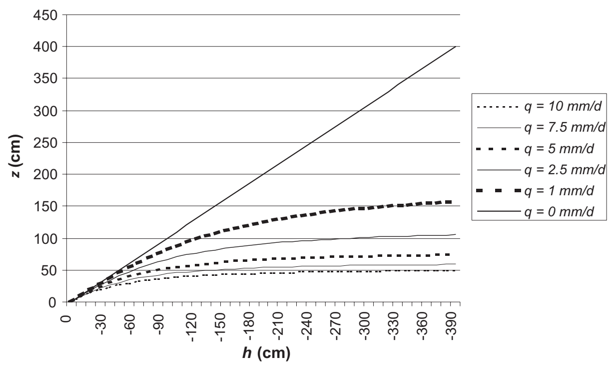
$z_{1,2}$ = vertical coordinates (positive upward and $z = 0$ at groundwater table) (m);

$h_{1,2}$ = soil pressure heads (m); and

$K(\theta)$ = unsaturated hydraulic conductivity (m/d).

$K(\bar{\theta})$ is the hydraulic conductivity for $\bar{h}$ which is $(h_1 + h_2)/2$. With this equation, the soil pressure head profiles for stationary capillary rise fluxes can be calculated. From these pressure head profiles, the contribution of the capillary rise under shallow groundwater table management can be estimated. Figure 31 shows the capillary fluxes for a silty soil where the reference level is the groundwater table ($z = 0$).

FIGURE 31
Pressure head profiles for a silty soil for stationary capillary rise fluxes



Box 5 provides an example on how Figure 31 can be used to estimate maximum capillary rise.

BOX 5: CAPILLARY FLUX IN THE DRAINAGE PILOT STUDY AREA

In the drainage pilot study area (previous calculation examples), maximum crop evapotranspiration is 7.5 mm/d. If it is assumed that the soil moisture is readily available up to a soil pressure head of 400 cm, a water table depth up to 60 cm below the bottom of the rootzone can deliver 100 percent of the crop water requirements through capillary rise (Figure 31). If the water table is not recharged, the water table depth will increase and therefore the capillary flux will decrease. For example, at a depth of 160 cm below the rootzone the capillary rise into the rootzone is reduced to 1 mm/d.

Annex 5 provides more theoretical background and an example to show the calculation procedures to derive the soil pressure head profiles for a given stationary capillary flux.

Instead of calculating the capillary rise manually, computer programs might be used. Programs are available based on both steady-state and non-steady-state models. An example of a steady-state model is CAPSEV (Wesseling, 1991). Non-steady-state models include SWAP (Kroes *et al.*, 1999). SWAP is not specifically designed to calculate capillary rise but mainly to simulate water and solute transport in the unsaturated zone and plant growth.

Maintaining a favourable salt balance under shallow water table management

The salt balance and salt storage equations apply for situations where the downward flux dominates over the capillary rise. When shallow groundwater table management is implemented, capillary rise is induced and the upward flux dominates over the downward flux. Where no irrigation and percolation occurs during the calculation period, the change in salt storage in the rootzone equals:

$$\Delta S = GEC_{gw} \tag{18}$$

The change in salt storage is a function of the amount of capillary rise (as calculated in the previous section) and the salinity of the groundwater. The accumulated salts in the rootzone need to be leached before the next growing season. In arid and semi-arid climates where rainfall is absent during this period, it is important to calculate the amount of water required to leach the accumulated salts from the rootzone.

Assuming that the amount of evaporation can be ignored during the leaching process and that the soil moisture has been restored to field capacity so that the leaching efficiency coefficient f_i equals f_r, the depth of leaching water can be calculated if the leaching coefficients are known. Under the assumption that there is no rainfall, the following equation can be used to calculate the depth of leaching water required (Van Hoorn and Van Alphen, 1994):

$$fqt = W_{fc} \ \ln \frac{2EC_{eo}\text{-}EC_I}{2EC_{et}\text{-}EC_I} \tag{19}$$

where:

f	= leaching efficiency coefficient (-);
q	= vertical flow rate (mm/d);
t	= time required to leach the accumulated salts (d);
EC_{eo}	= salinity in the rootzone at the start of the calculation period (dS/m); and
EC_{et}	= the desired salinity in the rootzone at the end of the period (dS/m).

Calculation example of the impact of shallow water table management on salinity buildup and leaching requirement

In the drainage pilot study area, the water table depth is normally high in the winter, crop evapotranspiration is low and irrigation water supplies are limited during the winter season. It is ideal to practise shallow groundwater table management in the winter when wheat is the main crop. Total crop evapotranspiration for wheat is 385 mm. The total seepage inflow during the wheat growing season is 0.5 mm/d * 150 d = 75 mm. The salinity of the seepage inflow is 5 dS/m and the salinity of the irrigation water is 1.0 dS/m. The water table depth (*WT*) at the start of the season is 0.7 m. The readily available soil moisture for the silty soil as a volume percentage (θ) is 0.191. The drainable pore space (μ) 0.15. The soil moisture content in the rootzone at field capacity (θ_{fc}) is 0.36.

It is assumed that all farmers in the area will practice shallow groundwater table management and no additional seepage into the area is induced. Under these assumptions the following steps need to be followed to calculate the change in groundwater table depth and the contribution of the capillary rise ($G = qt$) towards crop evapotranspiration:

Step 1. Calculate z, which is WT - half of the depth of the rootzone (D_{rz}).

Step 2. Estimate the initial equilibrium θ (Table 3, Annex 5) for the calculated z. If the change in soil moisture content (ΔW) = 0 then θ at the start of the next calculation period is equivalent to the equilibrium θ for the calculated z. At equilibrium conditions $h = z$. Otherwise, θ = equilibrium θ - ($\Delta W / D_{rz}$). If θ drops below the readily available soil moisture irrigation is required.

Step 3. Estimate the maximum q by using Figure 31.

Step 4. Calculate maximum G which is maximum qt.

Step 5. Check if the maximum G can cover the water uptake by crops. If maximum G minus ET_{crop} is positive then the actual G is equivalent to ET_{crop}. Otherwise, the actual G is equivalent to the maximum G.

Step 6. Calculate the change in soil moisture content (ΔW) = (ET_{crop} - actual G).

Step 7. Calculate the drop in groundwater table (Δh) = ((actual G - S_i) / μ).

Step 8. The WT in the succeeding month is the WT of the calculated month plus Δh.

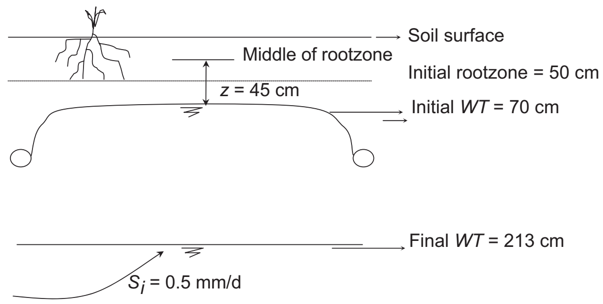
Step 9. Repeat previous steps for all the months in the calculation period.

The below matrix presents a summary of the calculations of the changes in groundwater table depth and the contribution of the capillary flow towards crop evapotranspiration.

	Dec	Jan	Feb	Mar	Apr	Total
WT (cm)	70	89	116	173	203	213
D_{rz} (cm)	50	80	100	100	100	
z (cm)	45	49	66	123	153	
θ (for $h = z$)	0.42	0.42	0.41	0.36	0.31	
maximum q (cm/d)	> 1.0	> 1.0	0.75	0.2	0.1	
maximum G (cm)	> 30	> 30	22.5	6	3	> 91.5
S_i (cm)	1.5	1.5	1.5	1.5	1.5	7.5
ET_{crop} (cm)	4.3	5.5	10.1	11.0	7.6	38.5
actual G (cm)	4.3	5.5	10.1	6.0	3.0	28.9
ΔW (cm)	0	0	0	- 5.0	- 4.6	- 9.6
Δh (cm)	19	27	57	30	10	143

The calculation example shows that the water table drops from 70 cm to 213 cm below the soil surface (Figure 32). The soil moisture content decreases towards the end of the growing season but does not get below the readily available soil moisture. Irrigation is therefore not required.

FIGURE 32
Change in water table during the winter season in the drainage pilot study area



Assuming that the shallow groundwater salinity is equivalent to the salinity of the seepage inflow into the area, the change in salt storage in the rootzone over the winter season is:

ΔS = 289 mm * 5 dS/m = 1 445 ECmm

If the salinity (EC_e) at the start of the season was 2 dS/m, the salinity at the end of the season would be:

S_{end} = S_{start} + ΔS

S_{end} = 2 * 2 * 360 + 1 445 = 2 885 ECmm

$EC_{e\ end}$ = 2 885 / (2 * 360) = 4.0 dS/m

Equation 19 can be used to calculate the leaching requirements. If it is assumed that the soil salinity should be lowered to 2 dS/m again before the sowing of cotton, the total leaching requirement is:

$$0.9\ qt = 360 \ln \frac{2*4.0-1.0}{2*2.0-1.0}$$

$$qt = 339 \text{ mm}$$

This leaching might be obtained during normal irrigation during the following season and may not need any special leaching irrigation unless the salts have actually accumulated in the seed zone.

LAND RETIREMENT

Land retirement involves taking land out of production because of water shortage and/or soil and water quality considerations. Elevated concentrations of toxic trace elements or extremely high salt concentrations in combination with waterlogging problems may be reasons for taking land out of production. By retiring salt-affected land, the total salt or toxic trace element load to be disposed of is reduced. Moreover, land retirement conserves irrigation water for reallocation to other beneficial uses such as public water supply, wetlands, etc. However, in California, the United States of America, water districts would generally prefer to retain the conserved water for application to other lands within the district preferably on non-problem soils. In water short regions of the Coleambally area in Australia, there is a policy to retain a minimum amount of water (2 000 m^3/ha) to keep retired lands from becoming salinized so that retired lands do not become a discharge zone (Christen, E.W., personal communication, 2001). If problem soils are encountered in the planning stages of new irrigation projects, it might be decided to leave these uncultivated. Excluding land during the planning stages is comparable to land retirement in developed projects. Where large contiguous blocks of lands are retired, native vegetation and wildlife can be sustained.

Hydrologic, soil and biologic considerations

Land retirement appears to be an attractive solution to the drainage problem. However, analysis of hydrologic, biologic and soil consequences indicates land retirement is complex. For example, in the western side of the San Joaquin Valley (SJVDIP, 1999c), the waterlogged areas with high-selenium shallow groundwater are located downslope in the lower part of the alluvial fans and in the trough of the valley. The water table in upslope locations is deep and lateral subsurface flow of deep percolation from upslope areas contributes to downslope drainage problems. Simulations with a two-dimensional hydrologic model (Purkey, 1998) ascertained the following:

- Retirement of large contiguous lands both upslope and downslope would provide the greatest overall reduction in drainage volume and lowering of regional water table.
- Retirement of downslope parcels with an existing high water table would reduce drainage volume to a greater extent than retirement of the same area of upslope land over the short run.
- Retirement of undrained, upslope parcels would provide significant reduction in drainage from land immediately downslope and provide the greatest long-term relief.
- Retirement of downslope parcels does not prevent the long-term upslope expansion of the zone of shallow groundwater and the retired downslope parcels would be degraded by soil waterlogging and salinization.

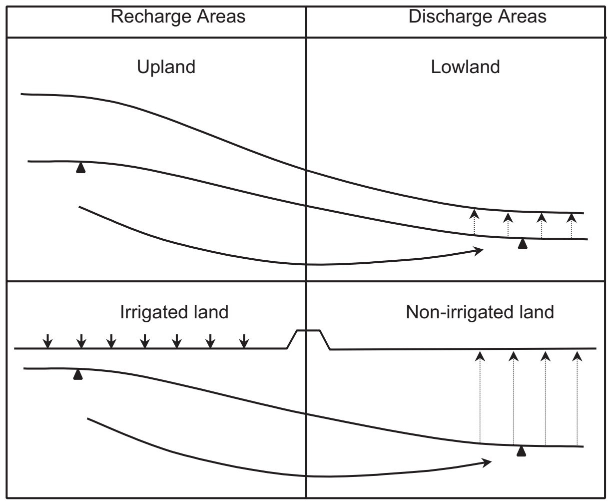
A land retirement programme can lead to many wildlife benefits by reducing the load of toxic trace elements and salts discharged into the environment, and especially if there is connectivity between retired land parcels for the restoration of native animals and plants. However, land retirement may potentially result in negative effects such as upwelling of saline shallow groundwater leading to excessive accumulation of trace elements and salts on the land surface, and the establishment of undesirable weed plant communities (SJVDIP, 1999c). Excessive salt accumulation might result in little or no vegetation cover, and wind-blown salt and selenium problems downwind. Therefore, retired lands require land, vegetation and water management in order to obtain wildlife benefits.

Selection of lands to retire

The potential reduction in drainage water effluent, salinity and toxic trace elements are the main criteria for selection of lands to retire. For example, in the San Joaquin Valley land areas generating selenium concentrations of more than 200 ppb are prime candidates for land retirement, and those generating more than 50 ppb are secondary candidates for land retirement. Where reduction of drainage effluent is the major goal, the retirement of undrained upslope land would provide the greatest long-term relief while retirement of downslope waterlogged land would reduce the drainage volume to a greater extent on a short-term basis.

One of the major concerns in land retirement is the accumulation of salts and trace elements at or near the soil surface destroying the vegetation cover and thus endangering its long-term sustainability. In this respect, it is important to make a distinction between recharge and discharge areas (Figure 33).

Figure 33
Recharge and discharge areas



It is relatively easy to differentiate between recharge and discharge areas in hilly and undulating terrain. Recharge occurs in the uplands and discharge in the lowlands and valley bottoms. In irrigated lands, recharge and discharge areas are more difficult to identify. Before an irrigation event in waterlogged lands, the land becomes a discharge area under the influence of evapotranspiration while just after irrigation the land turns into a recharge area. For more long-term trends, it would be desirable to ascertain directions of groundwater flow pathways for the identification of problem sites. Retirement of irrigated land and re-establishment of natural vegetation cover in irrigated flat plains will result in the creation of localized discharge areas.

Experience from Australia (Heuperman, 2000) shows that re-vegetation with salt tolerant crops in recharge areas with high groundwater tables is sustainable from a salinity point of view. It also reduces the recharge to the groundwater, thereby relieving lateral inflow in discharge areas. The plants in these areas rely on surface water input for evapotranspiration, which is either rainfall or irrigation. Land retirement in discharge areas with water tables within the critical water table depth combined with the introduction of (natural) vegetation will lead to the accumulation of salts in the vegetation's rootzone. Land retirement under these conditions is not sustainable if accumulated salts are not flushed down occasionally. Adequate drainage is required for the removal of saline drainage effluent.

Management of retired lands

The management of retired lands differs for upland and lowland conditions. In the uplands, the water table is deep and the vegetation depends on surface water input to meet the crop water requirement. In lowland areas, a shallow saline groundwater table normally lies beneath the retired land. Maintenance of the water and salt balance for both uplands and lowlands depends on the water requirement and salt tolerance of the vegetation. Annex 6 presents salt tolerance data for trees and shrubs that are potentially suitable for retired lands.

It is sometimes suggested that the planting of salt tolerant trees, shrubs and fodder crops may help to attain a positive salt balance. Chemical analysis of the bulk of dry weight of crops showed that 5 percent of the dry weight of a plant consists of mineral constituents (Palladin, 1992; and Pandey and Sinha, 1995). Kapoor (2000) assumes that these mineral constituents are derived from the soil solution. In this case, the biomass produced by salt tolerant crops on a yearly basis per hectare multiplied by the percentage mineral content of the plant is the total annual salt extraction per hectare of salt tolerant vegetation. The calculation example provided by Kapoor (2000) shows a favourable salt balance for an area where the only source of salt is imported surface water with a low salt concentration (86 mg/litre). However, from studies carried out in California and Australia where salinity levels are considerably higher there seems to be no real evidence that salt tolerant crops remove substantial amounts of salts from the soil water solution (Heuperman, 2000; Chauhan, 2000). Therefore, the water and salt balance equations as used for conventional agricultural crops also apply to maintaining favourable water and salt balances for vegetation tolerant to saline and waterlogged conditions that is used on retired land.

Chapter 6

Drainage water reuse

INTRODUCTION

In areas where irrigation water is scarce, the use of drainage water is an important strategy for supplementing water resources. Furthermore, reuse may help alleviate drainage disposal problems by reducing the volume of drainage water involved. The reuse of drainage water for irrigation can reduce the overall problems of water pollution. Reuse measures consist of: reuse in conventional agriculture; reuse to grow salt tolerant crops; IFDM systems; reuse in wildlife habitats and wetlands; and reuse for initial reclamation of salt-affected lands. This chapter deals with only those drainage water reuse measures that relate to agricultural production.

Drainage water is normally of inferior quality compared to the original irrigation water. Adequate attention needs to be paid to management measures to minimize long-term and short-term harmful effects on crop production, soil productivity and water quality at project or basin scale. The drainage water quality determines which crops can be irrigated. Highly saline drainage water cannot be used to irrigate salt sensitive crops, but it can be used on salt tolerant crops, trees, bushes and fodder crops. A major concern in reuse measures is that drainage water from reused waters is often highly concentrated, requiring careful management.

RELEVANT FACTORS

Drainage water quality is the major concern in reuse possibilities as it defines which crops can be irrigated and whether long-term degradation of soil productivity is a major issue. On the other hand, the soil type, drainage conditions of the land and the crop salt tolerance define what quality drainage water can be used for irrigation in combination with the availability of other freshwater resources.

Pollutants from surface runoff, i.e. sediments, pesticides and nutrients, play a minor role in reuse for crop production. However, for sustainable agricultural practices and to prevent environmental degradation, nutrients supplied with reused drainage water should be deducted from the fertilizer requirements in order to prevent imbalanced and excessive fertilizer application.

Subsurface drainage water generally shows increased concentrations of salts and sometimes certain trace elements and soluble nutrients. Salts and trace elements play a major role in the reuse of drainage water. Above a certain threshold value, high total concentrations of salts are harmful to crop growth, while individual salts can disturb nutrient uptake or be toxic to plants. A high sodium to calcium plus magnesium concentration ratio may cause unstable soil structure. Soils with unstable structure are subject to crusting and compaction, degrading soil conditions for optimal crop growth. Toxic trace elements such as boron can interfere with optimal crop growth and others such as selenium and arsenic can enter the food chain when crops are irrigated with water containing high concentrations of these trace elements. This is of major concern for human and animal health.

Considerations on the extent of reuse

Table 13
Quality indicators for some main drains

Indicator \ Drain	Upper Serw	Hamul	Upper No.1	Edko	Limits set in Law 48
TDS (mg/litre)	1395	1348	717	1075	-
BOD (mg/litre)	25	34	32	54	< 10
COD (mg/litre)	118	101	133	250	< 15
NH_4 (mg/litre)	3.0	14.4	27.9	1.1	< 0.5
MPN (10^6/100ml)	1.2	480	0.2	0.2	< 0.005
TSS (mg/litre)	251	170	202	450	-

Source: DRI, 1997a.

Municipalities and industries often use agricultural main drains to dispose of their wastewater. In many areas around the world, municipal and industrial wastewater is either insufficiently treated, or not treated at all. Bacteriological and organic and inorganic compounds seriously pollute main drains, posing an environmental hazard to both human and wildlife and restricting reuse from main drains. For example, in the Nile Delta, Egypt, the discharge of untreated wastewater is a major concern for the sustainability of an irrigated agriculture that depends heavily on the large-scale reuse of agricultural drainage water to meet water shortfalls. Table 13 lists several water quality indicators of some of the main drains and the official limits for the quality of drainage water that is allowed to be mixed with freshwater resources. The biochemical oxygen demand (*BOD*) is defined as the amount of oxygen consumed by microbes in decomposing carbonaceous organic matter. The chemical oxygen demand (*COD*) is the amount of oxygen required to oxidize the organic matter and other reduced compounds. The high chemical versus biological oxygen demand (*COD*/*BOD*) ratios imply significant industrial pollution (EPIQ Water Policy Team, 1998). Total coliform is the most probable number of faecal coliform (*MPN*) in 100 ml and a high value indicates severe pollution from municipal sewage water.

The degraded water quality threatens the expansion and even the continuation of the reuse of drainage water from the main drains (official reuse) in the Nile Delta. Since the 1990s, many reuse mixing stations have been under increasing pressure of water quality deterioration. Indeed, since 1992, 7 of the 23 main reuse mixing stations have been entirely or periodically closed (DRI, 1995). Due to the increasing deterioration of water quality, new opportunities for reuse of drainage water are being explored. Between the centralized official reuse and the localized unofficial reuse (where farmers pump drainage water from the collector drains directly onto fields), there is the option to capture drainage water from branch drains and pump it into the branch canals at their intersections. This level of reuse, termed intermediate reuse, captures relatively good quality drainage water before it is discharged into the main drains where it is lost for irrigation because of pollution from other sectors. This problem highlights the need for catchment level planning to protect and use all water resources sustainably.

Maintaining favourable salt and ion balances and soil conditions

Maintaining a favourable salt balance

The major concern in the reuse of agricultural drainage water is the buildup of salts and other trace elements in the rootzone to such an extent that it interferes with optimal crop growth and degradation of the aquifers. Applying more water than necessary during the growing season for evapotranspiration can leach the salts. In areas with insufficient natural drainage, leaching water will need to be removed through artificial drainage. As safe disposal of agricultural drainage water is often the hindrance to sustainable drainage water management, the solution in reuse lies in applying just enough water to maintain a favourable salt balance. This was termed in Chapter 5 as beneficial non-consumptive use.

Where the crop tolerance to salinity and the salinity of the irrigation water are known, the *LR* can be calculated (Chapter 5 and Annex 4). The following example uses Equations 12-15 presented in Annex 4 to calculate the *LR* for lettuce, a salt sensitive crop with an EC_{ts} of 1.3 dS/m. The EC_i of the applied water, a mixture of freshwater and drainage water, is assumed to be 1.2 dS/m and the ET_{crop} is 430 mm, and no rainfall occurs during the growing season. The crop is grown on a cracking soil with an f_i of 0.8. From Equation 14, the LR_i is 0.23. The amount of applied water obtained from Equation 15 is 698 mm. If lettuce is irrigated under furrow irrigation, a safe margin of 1.2 to account for non-uniformity (Table 9) is incorporated. The total amount of applied water is then 838 mm, of which 670 mm mixed with the soil solution and 168 mm bypassed through the cracks.

The following example from the Broadview Water District in the San Joaquin Valley illustrates that drainage water reuse is a viable management option but that upper limits need to be established. Moreover, a minimum amount of leaching is required to avoid deleterious impacts on the more salt sensitive crops. The Broadview Water District is a landlocked tract of 4 100 ha of irrigated land without a surface drainage outlet since 1956. Surface irrigation return flow, both tailwater and subsurface drainage, was recycled back completely into the irrigation supply ditch. Table 14 contains a summary of water quality and mass flow of salts for this type of reuse. The annual 18 324 million m^3 of imported canal water was a low-salt, low-boron water while the captured annual 16 363 million m^3 of drainage water was a moderately saline, 2 mg/litre boron water. The blended water serving as the supply water to the Broadview Water District was considerably degraded. In terms of mass of salts, the captured drainage water contributed about 66 percent of the salts applied in the water district.

TABLE 14
Flow-weighted concentration of salinity and boron concentrations and mass transfer of salts in Broadview Water District, 1976

Description	*EC* (dS/m)	*TDS* (mg/litre)	B (mg/litre)	*TDS* (tonne/ year)
Fresh canal water	0.41	272	0.19	5 500
Captured drainwater	2.99	2 085	2.19	37 630
Mixed supply water	2.19	1 485	1.55	56 810*

* Water quality was measured weekly and the flows of fresh canal water and mixed supply water were measured daily while that of captured drainwater was estimated by monthly electrical charges on the pump and assumed pump efficiency. Thus, the mass of salts in captured drainwater appears to be underestimated.

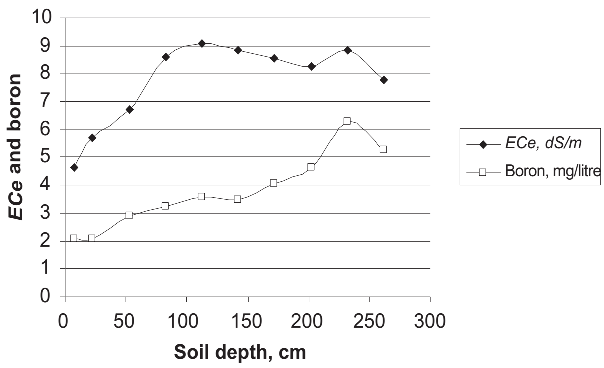
Source: Tanji, *et al.*1977.

Figure 34 shows the average concentration of soil salinity and boron in seven soil profiles in the Broadview Water District in 1976. The subsurface drains are installed about 2.4 m deep and spaced about 90 m apart. The crops grown in the district were cotton, tomatoes, barley, wheat, sugar beet, and alfalfa seeds. Over time, the 800 ha of tomato plantings dropped to 0 ha in 1987 when the seeds progressively failed to germinate. In 1989, the district gained access to discharge part of the saline drainage waters out of the district and reduce the blending of drainage water into the supply water. Within a few years, the tomato plantings were re-established at former levels.

Maintaining favourable soil structure

The sodium hazards of irrigation water are related to the ability of excessive sodium or extremely low salinity concentrations to destabilize soil structure. The primary processes responsible for soil degradation are swelling and clay dispersion. Provided that the salt concentration in the soil water is below a critical flocculation concentration, clays will disperse spontaneously at high exchangeable sodium percentage (*ESP*) values, whereas at low *ESP* levels inputs of energy are required for dispersion. The salt concentration in the soil water is crucial to determining soil physical behaviour because of its effects in promoting clay flocculation (Sumner, 1993). However, the boundary (*ESP* / salt concentration of the soil water) between stable and unstable conditions varies from one soil to the next and changes with the clay mineralogy, *pH*, soil texture, and clay, organic matter and oxide content (FAO, 1992b).

FIGURE 34
Average concentration of soil salinity and boron in seven soil profiles in Broadview Water District in 1976



The short-term effects of irrigating with water having excess sodium or very low salt concentrations relate mainly to infiltration problems. The reduction of infiltration can be attributed to the dispersion and migration of clay minerals into soil pores, the swelling of expandable clays and crust formation. The potential infiltration problems in relation to the quality of the irrigation water are normally evaluated on the basis of the salinity and *SAR*[1] of the irrigation water (Box 6).

BOX 6: ADJUSTED SODIUM ADSORPTION RATIO

The *SAR* is defined as $Na/((Ca + Mg)/2)^{1/2}$, in which the concentrations are expressed in milliequivalents per litre, and it is used to assess the infiltration problems due to an excess of sodium in relation to calcium and magnesium. It does not take into account the changes in solubility of calcium in the upper soil layers after and during irrigation. The solubility of calcium carbonate in the rootzone is influenced by dissolved carbon dioxide concentration, concentration of the solution and the presence of carbonates, bicarbonates and sulphates. FAO (1985b) has proposed a procedure for adjusting the calcium concentration of the irrigation water to the expected equilibrium value following irrigation.

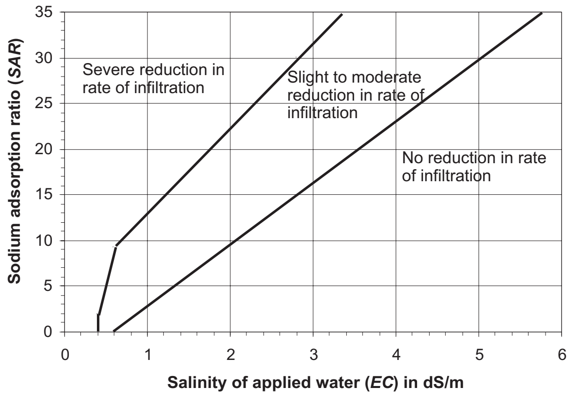
Various authors have developed stability lines related to the total salinity concentration and the *SAR*. The actual line that represents the division between stable and unstable soil conditions is unique for each soil type and varies with soil conditions. Published guidelines (Figure 35) on infiltration problems in relation to the *SAR* and the salinity of the applied water can therefore provide only approximate guidance. Where large-scale reuse of drainage water is planned and sodium hazards might be expected, stability lines need to be established for local conditions.

Infiltration problems caused by the sodicity of irrigation water also depend on irrigation and soil management. Especially at lower *SAR* levels when chemical bonding is weakened, but no spontaneous dispersion takes place, inputs of energy are required for actual dispersion. For example, sprinkler irrigation increases the likelihood of surface crusting due to the high physical disruption as the drops hit the soil surface aggregates. In soil management, incorporation of organic matter increases the stability of the soil aggregates and reduces the hazards of structural degradation as a result of sodicity.

[1] Because of the strong relation between the exchangeable sodium percentage (*ESP*) and sodium adsorption ratio (*SAR*) sodicity hazards are normally assessed through *SAR* of the soil or irrigation water because *SAR* is easier to determine than *ESP*.

Reduction of the hydraulic conductivity in the soil profile is normally a long-term effect resulting from the use of sodic water. In particular, the presence of carbonates and bicarbonates in the water could result in soil degradation in the long term because precipitation of calcium carbonate increases soil *SAR*. Calcium in the form of calcite is one of the first salts to precipitate. Upon further concentration magnesium salts will also precipitate.

FIGURE 35
Relative rate of water infiltration as affected by salinity and *SAR*



Source: FAO, 1985b.

There is little need to undertake action to increase infiltration and or hydraulic conductivity unless crop water or leaching requirements cannot be met or if secondary problems reduce crop yields or impede seedling emergence. Secondary problems include crusting of seed beds, excessive weed growth and surface water ponding that can cause root rot, diseases, nutritional disorders, poor aeration and poor germination (FAO, 1985b). Management options to mitigate and reduce these problems can be chemical, biological and physical. Chemical management options entail adding chemical amendments to soil or water and thereby changing the soil or water chemistry. The aim of biological methods is to improve soil structure or to influence the soil chemistry through the decomposition of organic materials. Physical methods include cultural practices to increase infiltration rates during irrigation and rainfall or to prevent direct contact between ponding water and plant stems, roots and seeds.

Chemical soil and water amendments to prevent infiltration problems

FAO (1985b) has analysed the management of infiltration problems by means of soil and water amendments in considerable detail. This section highlights the major issues. The aim of applying chemical amendments to soil or water is to improve poor infiltration caused by either a low salinity or by excessive sodium. The problem is most severe at low electrolyte concentration and high *SAR*. Improvements can be expected if the soluble calcium content is increased or a significant increase in salinity is achieved. Most soil and water amendments supply calcium directly or indirectly through acid that reacts with soil calcium carbonates. Acid is not effective where calcium carbonate is not present in the soil profile. However, calcium carbonate is often present in arid soils. Water amendments are most effective when infiltration problems are caused by low to moderate saline water (EC < 0.5 dS/m) with a high *SAR*. Where moderate to high saline water (EC > 1.0 dS/m) causes problems, soil amendments are more effective.

Amendments need to react quickly if they are to solve actively infiltration problems caused by irrigation water. Many of the amendments for improving or reclaiming sodic soils react only after oxidation. Thus, they are slow and less suitable for solving infiltration problems caused by irrigation water. Slow reacting amendments include acid forming substances such as sulphur, pyrite, certain fertilizers and pressmud from sugar-cane factories (in India and Pakistan).

Gypsum ($CaSO_4.2H_2O$) is the most commonly used amendment. It can either be applied to the soil or irrigation water. Infiltration problems normally occur primarily in the upper few centimetres of the soil. Hence, it is normally more effective to apply small frequent doses of gypsum left on the soil surface or mixed with the upper few centimetres of the topsoil than higher doses incorporated deeper in the soil profile.

> **Box 7: Conversion from meq/litre calcium to pure gypsum**
>
> 1 milliequivalent of calcium per litre = 86 mg of 100 percent gypsum per litre of water = 86 kg of 100 percent gypsum per 1 000 m^3 of water.

The application of gypsum to irrigation water to prevent infiltration problems usually requires less gypsum per hectare than when it is used as soil application. Gypsum application to water is particularly effective when added to low salinity water (< 0.5 dS/m) and less effective for higher salinity water because of the difficulty of obtaining sufficient calcium in solution. In practice, it is not possible to obtain more than 1-4 meq/litre of dissolved calcium, or 0.86-3.44 g/litre of pure gypsum (Box 7), in fast-flowing irrigation streams. In low salinity water, these small amounts of dissolved calcium ions may increase the infiltration rate by as much as 300 percent. To use gypsum as a water amendment, finely ground gypsum (< 0.25 mm in diameter) is preferred as it has a higher solubility. This is also normally the purer grade of gypsum. A drawback is that finely ground purer grades of gypsum are more expensive, which often prevents small farmers from using it. The coarser grinds and lower grades of gypsum are more satisfactory for soil application.

Experiments have been conducted with the placement of gypsum rocks in the watercourse. The problem is that the amount of calcium dissolving from the gypsum rocks is low so the effectiveness depends on the stream velocity and volume. Experiments from Pakistan have shown that sufficient head and length of the supply canal from the tubewell to the watercourse (which is also used for fresh canal water) are required to dissolve enough calcium in tubewell water. Where technically feasible, the use of gypsum stones has proved to be financially attractive for farmers (Chaudry *et al*., 1984). In large-scale applications, a drawback might be the cost of canal maintenance, as the gypsum blocks need to be removed before mechanical cleaning.

> 0.61 tonne of sulphuric acid = 1 tonne, 100 percent gypsum.

Sulphuric acid is an extensively used amendment for addressing infiltration problems. It is effective only where lime is present in the soil surface. This highly corrosive acid can be applied to the soil directly, where it reacts with lime making the naturally present calcium available for exchange with adsorbed sodium. Sulphuric acid reacts rapidly with soil lime making it a useful amendment to combat infiltration problems. When added to the irrigation water, it is neutralized by the carbonates and bicarbonates in the water and any excess would contribute towards dissolving the soil lime.

Biological soil amendments

Crop residues or other organic matter left in or added to the field improve water penetration. The more fibrous and less easily decomposable crop residues are more suitable for mitigating minor infiltration problems. Fibrous organic materials keep the soil porous by maintaining open voids and channels. The use of crop residues forming polysaccharides as a cementing agent is most effective where they are incorporated only in the upper few centimetres. Incorporation of easily decomposable organic matter and incorporation deeper into the soil profile do not generally help reduce infiltration problems although they do improve soil structure by producing polysaccharides that promote soil aggregation and enhance soil fertility. Another important aspect of the decomposition of organic matter is the production of carbon dioxide, which in turn increases the solubility of lime. Box 8 provides an example of a green manure from Pakistan and India.

Cultural practices

Cultivation is usually done for weed control or soil aeration purposes rather than to improve infiltration. However, where infiltration problems are severe, cultivation or tillage are helpful as they roughen the soil surface, which slows down the flow of water, so increasing the time during which the water can infiltrate. Cultivation is only a temporary solution. After one or two irrigations, another cultivation may be needed. Moreover, the construction of (broad) beds may help mitigate the ill effects of standing water as it prevents direct contact of the plants with the water. Research from Pakistan has shown that cotton in particular benefits from planting on broad beds.

BOX 8: THE USE OF *SESBANIA* AS A GREEN MANURE TO IMPROVE SOIL CHEMICAL AND PHYSICAL PROPERTIES

In India and Pakistan, farmers use *Sesbania* as a green manure to improve the chemical and physical properties of soils degraded by the use of sodic tubewell water. They also use it in the reclamation of alkali soils. *Sesbania* decomposes rapidly, producing organic acids which help to dissolve soil lime. The more fibrous stems of *Sesbania* help to maintain open voids and channels. In addition, *Sesbania* is a nitrogen-fixing tree and thus helps to improve soil fertility. The young branches of the tree can serve as fodder. Because of all these characteristics, *Sesbania* is a very popular crop especially among farmers using poor quality tubewell water in Punjab, Pakistan.

Maintaining favourable levels of ions and trace elements

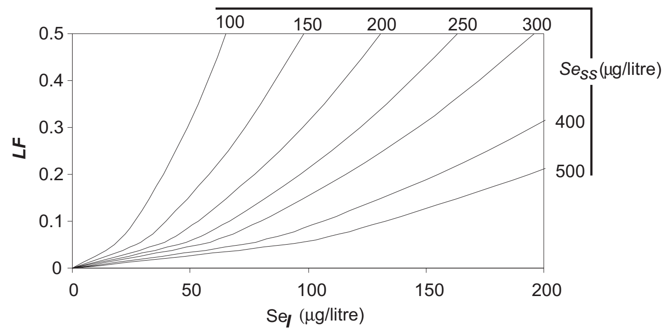
High concentrations of trace elements in soil, ground and drainage water can occur in association with high salinity and can be affected by the same processes. However, in some places they may also occur independently of salinity. In examining ways to control levels of ions and trace elements in the rootzone, it is necessary to understand the processes that affect their mobility. Deverel and Fujii (1990) provide a framework for evaluating concentrations of trace elements in soil and shallow groundwater. The two processes that largely control the mobility of trace elements in the soil water are: i) adsorption and desorption reactions; and ii) solid-phase precipitation and dissolution processes. These processes are influenced by changes in *pH*, redox state and reactions, chemical composition and solid-phase structural changes at the atomic level (Hinkle and Polette, 1999).

There are no well-tested and simple models for estimating changes in trace element concentration as a result of irrigation and drainage water management. Nor have irrigation water quality criteria for trace elements been established. However, guidelines have been developed for trace elements based on results from sand, solution and pot trials, field trials with chemicals and laboratory studies of chemical reactions. Pratt and Suarez (1990) list recommended maximum concentrations for 15 trace elements. These guidelines are designed to protect most sensitive crops and animals from toxicity where the most vulnerable soils are irrigated.

Selenium

Albasel *et al*. (1989) reviewed the quality criteria for trace elements in irrigation water compiled by the National Academy of Engineering in 1973. The recommendation for selenium was that the concentrations in irrigation water should not exceed 20 µg/litre. The guideline was recommended for all irrigation water on any land without consideration for soil texture, *pH*, plant species, climate and other water characteristics such as sulphate concentration. In the San Joaquin Valley, specific conditions indicated that a review of this guideline was required: i) selenium in the drainage water is in the form of selenate, which is not readily adsorbed onto soil particles and is thus readily leached; ii) water containing high selenium concentrations also has a high total salinity content and thus requires high leaching fractions to prevent salinity build up in the soil profile; and iii) water containing selenium has high concentrations of SO_4^- that greatly inhibit plant uptake of selenium. Albasel *et al.* (1989) used the concentration fractions for various *LF*s to convert the concentration of selenium in the irrigation water (Se_I) into selenium concentration in the soil solution (Se_{ss}). To derive the relationships shown in Figure 36, they assumed a water uptake pattern of 40-30-20-10 percent from the first to the fourth quarter of the rootzone. The concentration factors under this water uptake pattern are 5.56, 3.76, 2.58, 2.05, 1.74 and 1.53, respectively, for *LF* 0.05, 0.10, 0.20, 0.30, 0.40 and 0.50. To establish guidelines on the basis of these relationships, it is necessary to know the maximum selenium concentration in the soil solution (Se_{ssm}) at which the maximum concentration of selenium in the harvested product (Se_{hp}) will not be exceeded. Moreover, it is necessary that the *LF* be achievable.

FIGURE 36
Relationship between leaching fraction of the soil solution, selenium concentration of the irrigation water and the selenium concentration in the soil solution



Source: Albasel *et al.*, 1989.

Research with alfalfa, which is the most sensitive crop to selenium accumulation in relation to the use of the harvested product, showed that Se_{ssm} is 250 µg/litre without exceeding the Se_{hp} of 4-5 mg/kg if the saline irrigation water is dominated by sulphate. Assuming that the *LF* would be 0.2 or more, the Se_I could be 100 µg/litre (Albasel *et al.*, 1989). Where other crops are grown on soils different from those in the San Joaquin Valley, new guidelines will be necessary. These should be based on soil and water properties and the maximum Se_{hp} specific for those crops and the use of the harvested product.

Boron

Boron is an essential micronutrient for plants but it is toxic at concentrations only slightly above deficiency. The range of boron tolerance varies widely among crop plants. Salt sensitive crops such as citrus, fruit and nut trees are sensitive to boron while salt tolerant crops such as cotton, sugar beet and Sudan grass tolerate higher levels of boron (FAO, 1985b). Leaching of soil boron is more difficult than soluble salts such as chloride. This is because of the slow dissolution of boron minerals and desorption of boron adsorbed to oxides of iron and aluminium in the soil. Hoffman (1980) established a relationship to calculate the relative decrease of soluble boron in soils during reclamation:

(20)

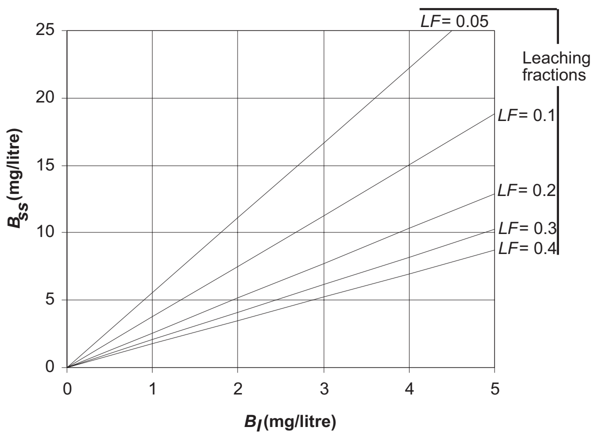
where:

B_{sst} = desired boron concentration in the soil solution (mg/litre);
B_{ss0} = initial boron concentration in the soil solution (mg/litre);
D_L = depth of leaching water (mm); and
D_s = depth of soil to be reclaimed (mm).

Keren *et al.* (1990) reported that native soil boron is more difficult to leach than boron accumulated from irrigation with boron-rich water.

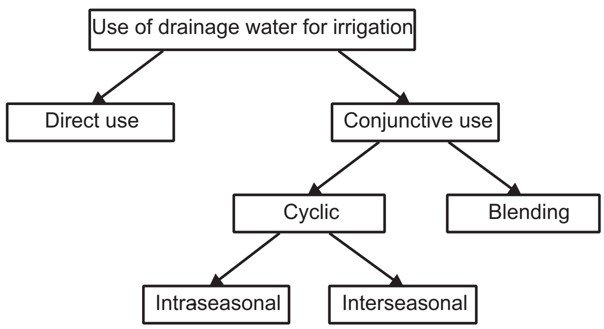
Where steady-state conditions exist between boron adsorbed and boron in the soil solution (B_{ss}), then the input and output of boron from the rootzone, and thus B_{ss}, is related to the boron concentration in the irrigation water (B_I) and the *LF* (Pratt and Suarez, 1990). To establish the relationship between B_I and B_{ss} (Figure 37), a water uptake pattern of 40-30-20-10 percent from the first to the fourth quarter of the rootzone is assumed. The concentration factors under this water uptake pattern are as the same as for selenium above. Lysimeter experiments have shown that Figure 37 can be used to assess the use of boron-containing water for irrigation (Pratt and Suarez, 1990) where near steady-state conditions exist.

FIGURE 37
Relationship between mean B_{ss} in the rootzone and between B_I for several leaching fractions



Source: Pratt and Suarez, 1990.

FIGURE 38
Use of drainage water for crop production



REUSE IN CONVENTIONAL CROP PRODUCTION

Drainage water of sufficiently good quality might be used directly for crop production. Otherwise, drainage water can be reused in conjunction with freshwater resources (Figure 38). Conjunctive use involves blending drainage water with freshwater. Alternatively, drainage water can be used cyclically with freshwater being applied separately. In cyclic use, the two water sources can be rotated within the cropping season (intraseasonal cyclic use), or the two water resources can be used separately over the seasons for different crops (interseasonal cyclic use). The choice of a certain reuse option depends largely on: drainage water quality; crop tolerance to salinity; and availability of freshwater resources. The quantity and time of availability of drainage water is of major importance. For example, where reuse takes place in an irrigation system in which water is distributed on a rotational basis, the probable mode of reuse is either direct or cyclic.

Direct use

The direct use of drainage water is implemented mainly at farm level, whereby the drainage water is not mixed with freshwater resources. Research results from India, Pakistan, Central Asia and Egypt (Part II), where surface irrigation methods are applied, show that drainage

Box 9: Direct reuse in Egypt and Pakistan

In the Nile Delta, Egypt, an official reuse policy exists whereby large-scale pumping stations mix the drainage water from the main drains into freshwater main canals. Unofficial direct reuse is generally a reaction of farmers to inadequate irrigation water supplies. Farmers directly pump the drainage water from the collector drain onto their fields without government approval. It is estimated that at least 3 000 million m^3 of drainage water is reused unofficially each year, almost equivalent to the volume of officially reused drainage water in 1995/96 (Egypt case study in Part II).

In Pakistan, another form of direct reuse takes place. In the 1960s, vertical drainage was initiated in the country to combat increasing waterlogging and salinity. As the surface irrigation system was initially based on low cropping intensities and low water allowances, farmers exploit drainage effluent from the tubewells on a large scale to augment insufficient surface water supplies. The drainage water is either used in conjunction with fresh surface water resources, or used directly for crop production. For farmers at the tailend of the water distribution systems, tubewell water is often the only source of water supply (Pakistan case study in Part II).

water can be used directly for irrigation purposes without severe crop yield reductions where the salinity of the drainage water does not exceed the threshold salinity value for the crops grown and good drainage conditions exist. As crops are often more sensitive to salinity during the initial growth stages, research in India has revealed the importance of pre-irrigation with good quality irrigation water. Higher crop yields were attained when freshwater pre-irrigation was applied with only drainage water being applied thereafter. Under these conditions, drainage water with salinity levels exceeding the threshold value could be used whilst maintaining acceptable crop yields.

In Pakistan, the major crops are wheat, cotton, sugar cane, rice and a variety of fodder crops. The water quality guidelines state that the maximum salinity of water used directly for irrigation is 2.4 dS/m (Pakistan case study in Part II). Assuming a concentration factor of 1.5[1] between irrigation water and saturated paste, this limit is rather high for sugar cane, which has a threshold value of 1.7 dS/m. Based on Maas and Grattan (1999), sugar cane irrigated with 2.4 dS/m water would yield 89 percent of the maximum potential yield. The long-term sustainability of direct use of drainage water depends on maintaining a favourable salt balance and preventing soil degradation due to sodicity problems. Box 9 describes direct reuse practices in Egypt and Pakistan.

Conjunctive use – blending

Where drainage water salinity exceeds the threshold values for optimal crop production, it can be mixed with other water resources to create a mixture of acceptable quality for the prevailing cropping patterns.

Where reuse takes place by mixing drainage water from main drains with surface water in main irrigation canals, the most salt sensitive crop determines the final water quality. For example, according to the regulations for the Nile Delta, the maximum salinity of the blended water is rather low to ensure optimal production of the major crops grown, i.e. cotton, maize, wheat, rice and berseem. Maize and berseem are the most salt sensitive crops with an EC_e threshold of 1.7 and 1.5 dS/m, respectively (Maas and Grattan, 1999). To ensure potential maximum yield

[1] In the guidelines for crop tolerance to salinity the relationship between soil salinity and salinity of the infiltrated water assumes a leaching fraction of 15 to 20 percent and a typical rootzone soil moisture extraction pattern of 40-30-20-10 percent from the upper to the lower quarters of the rootzone.

production of these two crops and assuming a concentration factor of two between the *EC* of the irrigation water and the EC_e, the maximum allowable salinity of the irrigation water is about 0.8-0.9 dS/m (Table 15).

TABLE 15
Drainage water quality criteria for irrigation purposes in the Nile Delta, Egypt

Salinity of drainage water (dS/m)	Restriction on use for irrigation
< 1.0	used directly for irrigation
1.0 - 2.3	mixed with canal water at ratio 1:1
2.3 - 4.6	mixed with canal water at ratio 1:2 or 1:3
> 4.6	not used for irrigation

Source: Abu-Zeid, 1988.

Where mixing takes place at the farm level, the salinity of the blended water can be adjusted towards the salt tolerance of individual crops. Table 16 shows an experiment from India, where blended water with different levels of salinity was used to cultivate wheat.

TABLE 16
Effect of diluted drainage water on wheat yield

EC_I, dS/m	Relative Yield (%)	
	All irrigations	Post-plant irrigations
0.5 (canal water)	100	100
3.0	90.2	-
6.0	80.4	95.8
9.0	72.5	90.3
12.0	56.4	83.7
18.0	-	78.0

Source: Sharma *et al.*, 2000.

This experiment shows that for the case of all irrigations applied with a blended water mixture having a maximum salinity of 3 dS/m, a potential crop yield of 90 percent of the maximum yield was obtained. According to the crop tolerance data published by Maas and Grattan (1999) and assuming a concentration factor of 1.5 between water salinity and soil salinity, the maximum potential yield could be attained. The difference between theory and the actual field situation might be caused by a discrepancy in the actual concentration factor, which is the inverse of the *LF*. Moreover, the actual yields depend on many factors including physical, climate and farm management practices. In general, where all irrigations use the blended water, the maximum allowable salinity depends on the threshold value of the crops. Where the pre-irrigation uses freshwater, the blended water could have a salinity level exceeding the threshold value without affecting yields. The experiment from India shows that water used in post-plant irrigations with a salinity level of three times (EC_I = 9 dS/m) the maximum allowable salinity level when only blended water is used (EC_I = 3 dS/m) results in 90 percent yields.

The leaching of salts is necessary to maintain a favourable salinity balance in the rootzone. The procedures presented above might be used for this purpose. On the other hand, if higher salinity levels are tolerated towards the end of the growing period or if a more salt sensitive crop is grown after a more salt tolerant crop, the salinity levels have to be reduced sufficiently low in order not to interfere with the growth of the next crop. Equation 19 could be used to calculate the required amount of leaching.

Conjunctive use – cyclic use

Cyclic use, also known as sequential application or rotational mode, is a technique that facilitates the conjunctive use of freshwater and saline drainage effluent. In this mode, saline drainage water replaces canal water in a predetermined sequence or cycle. Cyclic use is an option for where the salinity of the drainage water exceeds the salinity threshold value of the desired crop. A condition for cyclic use is that two different water sources can be applied to the field separately.

TABLE 17
Effect of cyclic irrigation with canal and drainage waters on yield of wheat and succeeding summer crops (t/ha)

Mode of application for wheat	Wheat	%	Pearl millet	%	Sorghum[4]	%
4 CW[1]	6.1	100	3.3	100	43.3	100
CW - DW[2] (alternate)	5.8	95	3.2	97	39.8	92
DW - CW (alternate)	5.6	92	3.2	97	39.5	91
2 CW - 2 DW	5.7	93	3.2	97	40.2	93
2 DW - 2 CW	5.4	89	-		39.5	91
1 CW - 3 DW	5.1	84	3.1	94	37.8	87
4 DW	4.5	74	2.8	85	34.1	77
Rainfall (mm)[3]	64		460		570	

[1] Canal water application, [2]drainage water application, [3] during the growing period, [4] as green forage
Source: Sharma *et al*., 1994.

Therefore, it is not normally applied at irrigation-scheme level but at a tertiary or farm level. In India and Pakistan, the canal irrigation water is delivered on a rotational basis to the watercourses (tertiary canal) and individual farms. This offers considerable potential for cyclic use on tertiary or farm level. Modelling and field studies have demonstrated the feasibility of the cyclic reuse strategy (Rhoades, 1987; Rhoades *et al*., 1988a and b; Rhoades 1989; and Rhoades *et al*., 1989).

The cyclic use of drainage water can be either intraseasonal or interseasonal. The latter mode of cyclic use follows the same principles for each cropping season as the direct use of drainage water.

In intraseasonal cyclic use, the strategy implies that non-saline water is used for salt sensitive cropping stages and saline water when the salt tolerance of the plant increases. Such experiments have been carried out in India (India case study in Part II). Saline drainage water (EC_i of 10.5-15.0 dS/m) was combined with canal water for use in a pearl millet/sorghum-wheat rotation. In the experiment, canal water was used for pre-plant irrigations and thereafter four irrigations each of 50 mm depth were applied as per the planned modes to irrigate wheat. Pearl millet and sorghum were given pre-plant irrigation and thereafter did not receive further irrigation except by monsoon rains during the growth period. Table 17 shows that the results of the experiment support the cyclic use strategy. No significant yield losses occurred in wheat when saline drainage water was substituted in alternate sequences (canal water - drainage water, or drainage water - canal water), or when the first two irrigations were with canal water and the remaining two irrigations were with drainage water, or when the first two irrigations were with saline drainage water and the remaining two irrigations were with canal water.

Intraseasonal cyclic use offers considerable potential as not all crops tolerate salinity equally well at different stages of their growth. Most crops are sensitive to salinity during emergence and early development. Moreover, the flowering stage is also critical. Tolerance to salinity generally increases with the age of the crop (Table 18). An exception is the salinity tolerance of mustard during the flowering to reproductive development stage.

The long-term sustainability of this drainage water management option entails devoting sufficient to maintaining a favourable salt balance and to preventing a buildup of trace elements in the rootzone to levels toxic to plant growth.

Cyclic use also requires attention to soil degradation as a result of using sodic water. A high exchangeable sodium percentage on the soil exchange complex does not normally lead to soil

TABLE 18
Crop response to salinity for three crops at various growth stages

Crop	Period	Response function	EC_0[1]	EC_{50}[2]
			(dS/m)	
Wheat	Average	$RY = 100 - 4.1\ (EC_e - 3.8)$	28.4	16.0
	Sowing time	$RY = 109.9 - 6.2\ EC_e$	17.3	9.7
	Mid season	$RY = 115.7 - 5.5\ EC_e$	21.0	11.9
	Harvest	$RY = 106.7 - 3.4\ EC_e$	31.1	16.7
Mustard	Average	$RY = 100 - 5.5\ (EC_e - 3.8)$	15.6	9.7
	Sowing time	$RY = 115.6 - 8.2\ EC_e$	14.1	8.0
	Mid-season	$RY = 168 - 12.6\ EC_e$	13.3	9.4
	Harvest	$RY = 106.6 - 3.3\ EC_e$	32.3	17.1
Greengram	Average	$RY = 100 - 20.7\ (EC_e - 1.2)$	6.6	4.2
	Sowing time	$RY = 115.3 - 20.9\ EC_e$	5.5	3.1
	Mid season	$RY = 150.3 - 28.5\ EC_e$	5.3	3.6
	Harvest	$RY = 157.2 - 24.8\ EC_e$	6.3	4.3

[1] $EC_0 = EC_e$ at which zero yield is obtained
[2] $EC_{50} = EC_e$ at which yields are reduced to 50 percent

Source: Minhas, 1998.

degradation if it is compensated by a high soil moisture salinity to suppress the extent of the so-called diffuse double layer. However, upon irrigation with low saline irrigation water or rainfall, the diffuse double layer swells resulting in soil dispersion. Adding sufficient soil or preferably water amendments to compensate for the high sodium to calcium and magnesium content in the saline irrigation water can prevent these problems.

CROP SUBSTITUTION AND REUSE FOR IRRIGATION OF SALT TOLERANT CROPS

Crop substitution

Crops differ significantly in their tolerance to concentrations of soluble salts in the rootzone. The difference between the tolerance of the least and the most sensitive crops may be tenfold. A number of salt tolerant crop plants are available for greater use of saline drainage effluent (Annex 1). Raising the extent of the salinity limits through selecting more salt tolerant crops enables greater use of saline drainage effluent and reduces the need for leaching and drainage. Table 19 shows promising cultivars from India and Pakistan. In other parts of the world, other salt tolerant varieties have been developed.

Reuse for irrigation of salt tolerant plants and halophytes

Where the irrigation water is too saline to grow conventional agricultural crops, irrigation of halophytes might be considered. The maximum amount and kind of salt that salt tolerant plants and halophytes can tolerate vary among species and varieties. Halophytes have a special feature as their growth is improved at low to moderate salinity levels (Goodin *et al.*, 1999). In contrast, salt tolerant crops have maximum growths up to a threshold salinity level after which growth is reduced (Figure 39). Salt tolerant plants and halophytes have been grown successfully in many places in the world to produce fuel, fodder and to a lesser extent food. Institutes in Australia have gathered useful salt tolerance data on a large number of trees and shrubs species (Annex 6). Growing salt tolerant plants and halophytes under saline conditions requires management of the

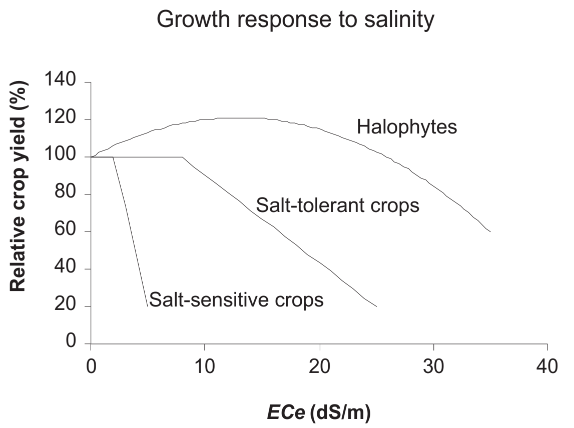
TABLE 19
Promising cultivars for saline and alkaline environments in India and Pakistan

Crop	Saline irrigation		Alkaline environment
	India	Pakistan	India
Wheat	Raj.2325, Raj. 3077,WH 157	LU26S, Blue Silver, SARC-1 (well-drained) Blue Silver, SARC-3, Pb-85 (waterlogged)	KRL1-4, KRL-1-19, HI 1077, WH 157
Pearlmillet	MH 269, MH 331		MH 2669, MH 280, MH 427
Mustard	CS416, CS 330-1, PUSA Bold	Gobi sarson	CS 15, CS52, Varuna
Cotton	DHY 286, G 17060	NIAB-78, MNH-93	HY 6
Sorghum	SPV-475, Spr 881, CH 511	Milo, JS-263, JS-1	SPV 475, CHI, CH 511
Barley	Ratna, RL 345, K169	PK-30064, PK-30130, PK-30132, PK-30316	DL 4, DL 106
Rice		NIAB-6, IRO-6, KS-282	

Source: AICRP Saline Water, 1998; and Qureshi, 1996.

salt balance in the rootzone. Where natural drainage is insufficient, artificial drainage is required to remove the leaching water.

FIGURE 39
Relative growth response to salinity of conventional versus halophytes



Source: Goodin *et al.*, 1989.

Fuel

Many people in developing countries rely on wood for cooking and heating. As agricultural land is required to feed growing populations, it is unlikely that good quality agricultural land will be used for fuel production (Goodin *et al*., 1999). Salt tolerant trees and shrubs can be grown for fuel production and building materials using saline water and marginal lands. Among the promising tree species for these purposes are *Prosopis, Eucalyptus, Casuarina, Rizophora, Melaleunca, Tamarix* and *Acacia*.

Fodder

Pasture improvement programmes in salt-affected regions throughout the world have used halophytes and salt tolerant shrubs and grass species. Trees and shrubs can be valuable complement to grasslands. They can serve as a nutrient pump and lower saline shallow groundwater tables. They are less susceptible to moisture deficits and temperature changes than grasses. They might also provide valuable complementary animal food or fuelwood. Salt tolerant grasses, shrubs and trees with potential for fodder use include:

Grasses	Shrubs	Trees
Kallar grass (*Leptochloa fusca*)	*Atriplex sp.*	*Acacia sp.*
Silt grass (*Paspalum vaginatum*)	*Mairiena sp.*	*Leucaena sp.*
Russian-thistle (*Salsola ibercia*)	*Samphire sp.*	*Prosopis sp.*
Salt grasses (*Distichlis spicata*)		
Channel millet (*Enchinochloa turnerana*)		
Cord grasses (*Spatina g.*)		
Rhodes grass (*Chloris gayana*)		

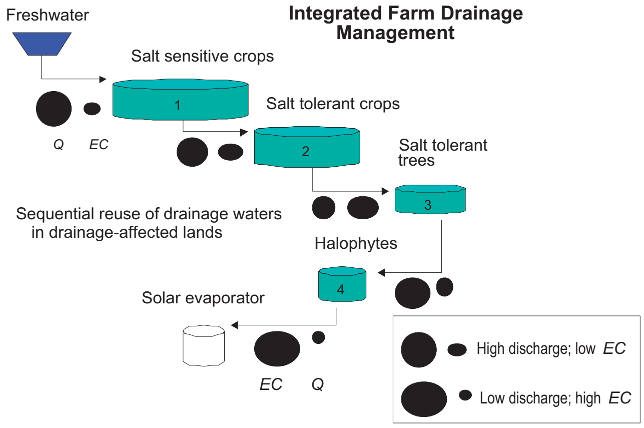
Other products

Salt tolerant plants can also produce other economically important materials, e.g. essential oils, gums, oils, resins, pulp and fibre. Moreover, salt tolerant plants can be used for landscape and ornamental purposes and irrigated with saline water, thereby conserving freshwater for other purposes (Goodin *et al*., 1999).

REUSE IN IFDM SYSTEMS

The IFDM system aims to utilize drainage water as a resource to produce marketable crops and to reduce the volume of drainage water to be discharged (SJVDIP, 1999d; and Cervinka *et al*., 2001). Figure 40 depicts the principles of a typical IFDM system. Under IFDM, drainage water is used sequentially to irrigate crops, trees and halophytes with progressively increasing salt tolerance. Each time the drainage water is reused, the volume of effluent is reduced and the salinity concentration increased. A typical IFDM system consists of four zones. In Zone 1, traditional salt sensitive crops are grown, e.g. vegetables, fruits, beans and corn. In Zone 2, traditional salt tolerant crops are grown, e.g. cotton, sorghum and wheat. In Zone 3, salt tolerant trees and shrubs are grown. In Zone 4, only halophytes can be planted. The final non-re-usable drainage water is discharged in a solar evaporator.

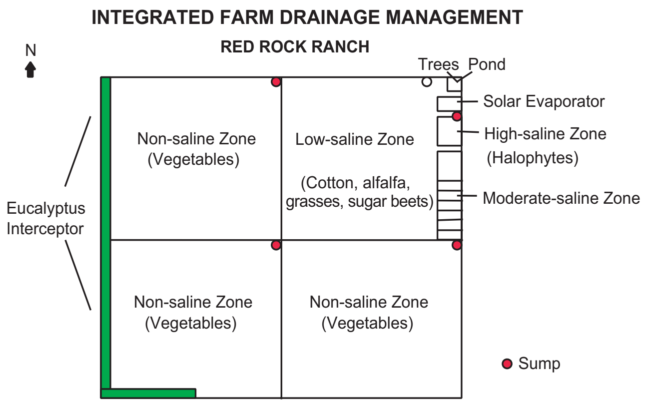
FIGURE 40
Principles of an IFDM system



Source: SJVDIP, 1999d.

The solar evaporator consists of a levelled area lined with plastic on which the brine is disposed and the crystallized salts are collected. The daily discharge of drainage water corresponds to the daily evaporation, this to prevent water ponding that attracts waterbirds. This is only important where high concentrations of toxic trace elements are present in the drainage water, otherwise a normal evaporation basin can be used.

FIGURE 41
Layout of sequential reuse of subsurface drainage waters and salt harvest at Red Rock Ranch



Source: SJVDIP, 1999a.

Figure 41 shows an example of IFDM from California and depicts the current layout of the 260-ha Red Rock Ranch. The ranch was waterlogged and salt affected to such an extent that

TABLE 20
Changes in salinity and boron by depth at locations in the tile drained Red Rock Ranch

Field ID	Soil depth	1995		1996		1997	
	cm	*ECe* dS/m	Boron ppm	*ECe* dS/m	Boron ppm	*ECe* dS/m	Boron ppm
10NW	0-30	10.0	15	1.8	2	1.0	1
	30-60	11.3	16	6.8	7	4.1	4
	60-90	10.3	9	9.5	14	6.6	9
10SW	0-30	13.8	13	3.1	4	1.5	4
	30-60	10.7	11	7.5	7	8.4	12
	60-90	9.6	8	8.3	7	10.7	12
10NE	0-30	10.0	4	2.0	2	2.0	1
	30-60	4.4	5	6.9	6	4.5	3
	60-90	5.8	6	9.0	10	5.5	4
10SE	0-30	2.9	3	1.1	1	0.8	1
	30-60	6.1	4	3.7	3	1.8	2
	60-90	7.7	5	7.3	6	3.1	7

Source: Cervinka *et al*., 2001.

the farmer was unable to obtain economic crop yields. Therefore, the farmer transformed it into an agroforestry reuse system with final disposal in a solar evaporator.

The shallow groundwater flows northeast, and in some areas the water table was within less than 30 cm of the land surface. The farmer planted rows of Eucalyptus *camendulensis* in the upslope western boundary to intercept some of the lateral groundwater, and then tile drained the four parcels of land progressively, starting from the southwest parcel, then the northwest and southeast parcels, and finally the northeast parcel. Within a few years of subsurface drainage, the land was reclaimed to the point where alfalfa and broccoli were successfully grown in the non-saline parcels with imported irrigation water of 0.5 dS/m EC_I and 0.2 ppm boron. Cotton, sugar beets and salt tolerant grasses were grown successfully in the low-saline parcel using tile drainage water (EC_I 6-8 dS/m) from the three non-saline parcels. Because there are no opportunities for off-farm disposal of drainage waters in this subarea, the residual drainage water from this ranch is sequentially reused until no longer usable. A portion of the northeast parcel has been set aside to irrigate saltgrass (EC_I about 10-20 dS/m) in the moderately saline zone. The subsurface drainage water from saltgrass is used to irrigate *Salicornia*, and then its drainage water (EC > 30 dS/m) is disposed into a solar evaporator to harvest salts. A market for the salts is currently being sought.

Table 20 documents the extent of reclamation of the salt-affected soils on Red Rock Ranch principally due to the installation of subsurface drainage and growing salt tolerant cotton and sugar beets initially. Today, about 75 percent of the land has been reclaimed sufficiently to grow salt sensitive, high-cash-value vegetable crops. The remaining 25 percent of the land is devoted to drainage water reuse in salt tolerant plants and salt harvest.

Table 21 details the average quality of the supply water and sequentially reused drainage water from salt sensitive crops, salt tolerant crops and halophytes. The quality of irrigation water for salt tolerant crops has improved and the salinity of the irrigation water is now about 6 dS/m in contrast to the reported 9.4 dS/m. Some difficulties have been encountered in managing the collected drainage water, especially for reuse on moderately saline and halophyte plots due to their comparatively small area. Drainage water disposed into the solar evaporator is controlled closely to prevent ponding by adjusting sprinkler irrigation rates and timing to daily *ETo* values.

TABLE 21
Water quality of supply and reused drainage waters on Red Rock Ranch

	EC dS/m	Na ppm	SO4 ppm	Cl ppm	B ppm	Se ppb
Irrigating salt sensitive crops	0.5	64	28	89	0.2	ND
Irrigating salt tolerant crops	9.4	1 422	3 070	1 189	10	322
Irrigating halophytes	31.0	7 829	12 686	4 722	56	609
Discharge into solar evaporator	28.2	7 598	12 814	4 436	62	706

Source: Cervinka *et al.*, 2001.

The extremely high selenium concentration in the drainage water is of concern to wildlife biologists. This system was initially established about 10 years ago but the present configuration has been operating for about 3 years and thus its sustainability is not yet known.

RECLAMATION OF SALT-AFFECTED LAND

Sodic soils often have low hydraulic conductivity as a result of the high sodium percentage on the soil exchange complex. The reclamation of sodic soils requires that a divalent solute (mainly calcium) pass through the soil profile, replacing exchangeable sodium and leaching the desorbed sodium ions from the rootzone. Therefore, the rate at which sodic soils can be reclaimed depends on the water flow through the soil and the calcium concentration of the soil solution.

The application of leaching water with a high electrolyte concentration promotes flocculation of the soils and thus improves soil permeability. This expedites the reclamation process. Amendments need to be added in order to replace sodium with calcium ions on the soil exchange complex. Over time, less-saline water needs to replace the saline leaching water in order to lower the salinity levels sufficiently to establish a crop.

The use of saline drainage water to reclaim salt-affected soils is not a permanent solution for reducing drainage effluent disposal volumes. It is only a substitute for the use of good quality irrigation water for reclamation purposes. FAO (1988) has compiled a list of procedures and measures for the reclamation and management of saline and sodic soils.

Chapter 7
Drainage water disposal

Requirements for safe disposal

Drainage water management is normally concerned with reducing the amount of drainage water and with managing its disposal. However, this aim is more complex than it appears. Drainage is practised to maintain aeration in waterlogged rootzone and/or to leach excess soil salinity to sustain agricultural production. The drainage water generated must then be managed for reuse purposes where it is of suitable quality and finally discharged or disposed of. The discharge of drainage waters in watercourses may have impacts ranging from beneficial to deleterious. The disposal of drainage water into wetlands, lakes, rivers and coastal waters entails considerations about the quantity and quality allowable and indeed sometimes required to maintain desirable ecological conditions and functions of that given water body.

To sustain irrigated agriculture, the maintenance of favourable salinity levels is the major concern in assessing the minimum drainage disposal requirements. When aimed at maximizing source reduction or reuse of drainage water, minimum leaching is required to maintain favourable salt balances in the rootzone. The minimum *LF* depends on the salinity of the irrigation water and the salt tolerances of the crops grown. Box 10 shows an example from the Nile Delta, Egypt.

Box 10: Maximum reuse and minimum disposal of drainage water in the Nile Delta, Egypt, based on maintaining favourable salt balance

Monitoring data showed that in 1993-94, 12 500 million m^3 of drainage water were disposed and 3 400 million m^3 were officially reused for agriculture (DRI, 1995). The effective reuse was 3 000 million m^3 assuming an *LF* of 12 percent. Data analysis showed that 13 300 million m^3 of drainage water with a salinity level of less than 4.7 dS/m (the official maximum salinity level allowable for reuse) was theoretically available for reuse. The required *LF* for this salinity level was estimated to be 0.19 (EPIQ Water Policy Team, 1998). Therefore, the effectively re-usable quantity of drainage water for consumptive use would be 10 800 million m^3, leaving 4 700 million m^3 (12 500 + 3 000 million m^3 - 10 800 million m^3) of drainage effluent to be disposed.

Minimum disposal requirements also depend on the requirements of downstream water uses. For example, many inland fisheries are currently threatened because of increasing water pollution, degradation of aquatic habitats and excessive water abstraction (FAO, 1999a; and Barg *et al*., 1997). With the increase in environmental awareness, there is increasing pressure to preserve sufficient water for aquatic environments, habitats and biodiversity. For example, wetlands are considered to be among the world's most valuable ecological sites. They form the habitat for many species of plants and animals. Wetlands perform various other vital functions, e.g. water storage, flood mitigation and water purification. Wetlands also provide economic benefits, of which fisheries, recreation and tourism are among the most important. These functions, values and attributes can only be maintained if the ecological processes of wetlands continue functioning (Ramsar Convention Bureau, 2000). If the ecological processes in the wetlands depend on drainage water inflow and if the functions of the wetlands are to be sustained, the minimum drainage discharge will depend on the quality and quantity of drainage water required

Box 11: Minimum drainage discharge requirements for maintaining the freshwater functions of the Northern Lakes, Egypt

The Northern Lakes (Maruit, Edko, Burrulus and Manzala) are located adjacent to the Mediterranean Sea and they are no more than 2 m deep. The lakes are economically important as they support a large fishery and many fish farms. Lake fisheries produce 52 percent of the nation's total fish production and provide employment to 53 000 anglers. The lakes are also ecologically important as they support a large bird population and serve as a stopover for migratory birds. As the lakes are also the sink for a large part of the drainage and wastewater from the Nile Delta prior to outflow to the sea, the quality and quantity of the drainage water discharge can threaten these two main functions of the lakes. Lake Maruit is heavily polluted and suffers from eutrophication due to drain discharges carrying industrial and municipal waste. The surviving fish species are limited and provide only a non-commercial food source to local residents. As the lake would need large quantities of freshwater inflow to recover its ecology, the consensus is that this lake will continue its salinization process and eventually lose all fish production and its ecological value (EPIQ Water Policy Team, 1998). The other three lakes depend on drainage outflow to maintain their brackish water environment. Based on the assumption that Lake Maruit will loose its function as a freshwater lake for fish production, the Drainage Re-use Working Group of the EPIQ Water Policy Team conducted an analysis of the required minimum drainage water outflow to lakes Edko, Burrulus and Manzala. For continued fish production, the salinity levels in the lakes should be maintained between 5.5 and 6.25 dS/m. Lake Edko, Lake Manzala and Lake Burrulus have salinity levels of 2.1, 3.2 and 5.6 dS/m, respectively. To maintain maximum salinity levels, the drainage outflow to Lake Manzala and Lake Edko can be reduced to a maximum 50 percent of the present outflow level. However, the outflow to Lake Burrulus is already on the low side and cannot be reduced any further. The minimum drainage discharge based on sustained freshwater lake fisheries is 8 500 million m^3 per year. This is 3 800 million m^3 more than the reuse potential based on maintaining the salt balance in the Nile Delta for agricultural production. Hence, less water is available for reuse if the lake fisheries are to be maintained.

for maintaining the ecological processes. Box 11 shows how the minimum drainage discharge or maximum reuse potential for agriculture is estimated based on sustaining the fisheries in the coastal lakes of the Nile Delta, Egypt.

Disposal conditions

Depending on the location, hydrology and topography of the drainage basin (and the ecology and environmental conditions of receiving water bodies), drainage water might be disposed to open surface water bodies, e.g. rivers, lakes, outfall drains, and oceans. The oceans are often regarded as the safest and the final disposal site for agricultural drainage water. This is true unless drainage water is contaminated with sediments, nutrients and other pollutants and the disposal site is in the vicinity of fragile coastal ecosystems such as mangroves and coral reefs. Therefore, at the point of discharge, pollution may be of much concern. In general, oceans have significant dilution or assimilative capacity. However, this is also limited in many cases especially in enclosed and semi-enclosed seas. Inland drainage water disposal to freshwater bodies such as lakes and rivers requires care. Rivers are normally used for different water use purposes requiring certain qualities of water, while the accumulation of salts and other pollutants in freshwater lakes threatens ecosystem functions and aquatic life.

Disposal into rivers or oceans is not always possible. In closed drainage basins, alternative disposal options need to be sought. The options for closed hydrologic basins include evaporation ponds, deep-well injection and integrated drainage management systems. Drainage water management in closed basins presents numerous environmental and water quality challenges.

Disposal in wetlands offers much potential but at the same time there are also accompanying constraints. Wetlands might be either natural or specially constructed for drainage water reuse. In the former case, the maintenance of ecological functions depends on the quality and quantity of drainage water inflow.

Disposal in freshwater bodies

The main aim of safe disposal in freshwater bodies such as lakes and rivers is to protect beneficial downstream water uses. Rivers and lakes are normally multifunctional (municipal drinking-water, industrial water, fishing, recreation and agriculture) and have an intrinsic ecological value. Furthermore, rivers feed lakes, floodplains, wetlands, estuaries and bays. To protect these functions, it is necessary to determine the assimilative capacity of the river and, where necessary, of connected or receiving ecosystems. It is also necessary to identify the constituents of concern in the drainage water in order to determine its discharge requirements. Moreover, the effects on sediments, riparian habitats and floodplains warrant consideration.

The discharge requirements should specify the maximum allowable concentration of each constituent of concern and the volume of drainage water that will be acceptable. Concentration is the water quality parameter normally used in drinking-water standards or for the health of aquatic animals and plants. However, placing only concentration limits on discharge might encourage dilution and inefficient water use. Where both concentration and load limits (product of water concentration and water volume) are enforced, they tend to promote efficient water use. For example, in the San Joaquin Valley, the discharge limitations into the San Joaquin River include limits on concentration of specific constituents and their loads. The concentration limitations include limits on salinity, boron, selenium and molybdenum in order to protect downstream water quality for domestic and agricultural uses and to safeguard the environment. To discourage dilution or inefficient water usage, load limits for these constituents are enforced for the discharge of irrigation return flows into the San Joaquin River.

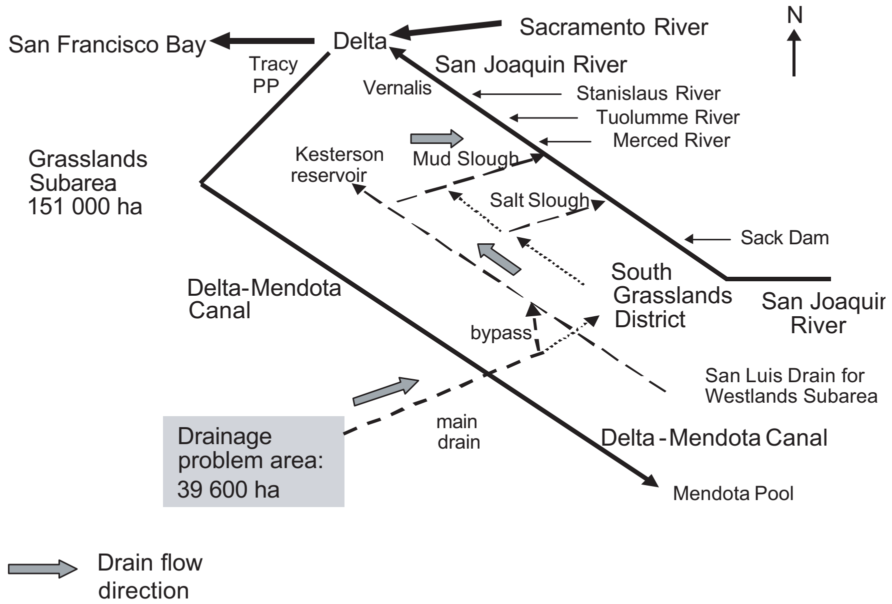
The assimilative capacity of the receiving water body varies from place to place and from time to time depending on numerous local conditions. These include climate and physical conditions and the upstream uses. The River Yamuna in Haryana, India, provides an example of seasonal differences in disposal opportunities. During the dry winter season, November-March, the flow of the River Yamuna is less than 50 m^3/s. From July to September, during the monsoon season, the flow exceeds 1 000 m^3/s with a peak of 1 150 m^3/s in August. The salinity of river water during monsoon is less than 0.2 dS/m. The river's high flow and low salinity during the monsoon period provides an opportunity for the disposal of saline effluents. As the water table shows a marked rise during the monsoon season, major pumping for drainage takes place between July and September. Table 22 shows projections of the amounts of subsurface drainage water that could be disposed into the River Yamuna. The criterion used was that the resultant salinity of the water in the river after mixing should be less than 0.75 dS/m. A significant drainable area

Table 22
Allowable subsurface drainage discharge and drainable area into the River Yamuna

Month	Allowable discharge (m^3/s)		Drainable area (ha)	
	Effluent salinity (dS/m)			
	6	10	6	10
June	0.9	0.5	5 000	3 000
July	25.4	14.4	146 000	83 000
August	47.6	27.0	274 000	156 000
September	6.5	3.7	37 000	21 000
October	3.0	1.7	17 000	10 000

Source: FAO, 1985a.

FIGURE 42
Map of the Grasslands subarea and drainage water discharge



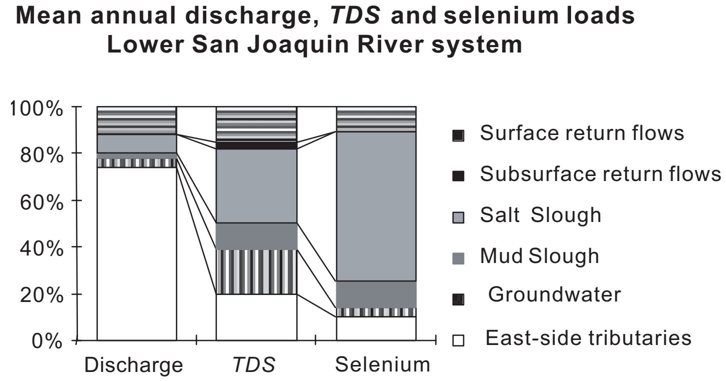
with salinity problems can use the River Yamuna as a possible drainage outlet between July and September (India case study in Part II).

Drainage from the Grassland subarea in the San Joaquin Valley provides another example of the possibilities and constraints of river disposal. Figure 42 presents a schematic map of the area (39 600 ha) that has opportunities to discharge its irrigation return flow into the San Joaquin River. However, there are some constraints as dictated by the waste discharge requirements set by the Central Valley Regional Water Quality Control Board (CVRWQCB, 1999).

In the recent past, irrigated agriculture discharged its irrigation return flow into the North and South Grasslands District, an entity devoted to wetland habitat for private duck clubs and birdwatching as well as pastureland for grazing animals. Since the discovery of selenium poisoning of waterbirds at Kesterson Reservoir from subsurface drainage waters conveyed from the Westlands Water District, drainage from the problem area is no longer discharged into the Grasslands Water District. Instead, it is discharged directly into the San Joaquin River at Mud Slough via the San Luis Drain and a bypass.

Figure 43 shows that although their flows are minor, Salt Slough and Mud Slough are major contributors of *TDS* and selenium in the lower reaches of the San Joaquin River. Thus, the CVRWQCB has placed monthly waste discharge requirements for Salt Slough and Mud Slough (CVRWQCB, 1998). The water quality objectives were established to protect downstream water uses as well as export water to southern California. The salinity water quality objective for Vernalis, a benchmark station on the river, is 1 dS/m for a 30-day running average from 1 September to 30 March, and 0.7 dS/m for a 30-day running average from 1 April to 31 August. The boron objective is placed from Sack Dam to Merced River, the reach above and below Salt Slough and Mud Slough, at 2.0 mg/litre monthly mean for the year with a maximum of 5.8 mg/litre for a given month. The selenium water quality objective is placed on Salt Slough and Mud Slough at 5 μg/litre, 4-day average, to protect aquatic biota and a maximum annual load of 3 632 kg to control mass emission. Exceeding the specified allowable loads of salt and selenium

FIGURE 43
The significance of irrigation return flows from Salt Slough and Mud Slough on the quality of the San Joaquin River system



Source: CVRWQCB, 2000.

results in monetary fines to the dischargers. The selenium load is the most difficult for irrigated agriculture to meet.

Generally, rivers continually cleanse themselves. In most cases, lakes do not have this capacity as they may not have an outlet, or flow volume from the lake is limited. The disposal of drainage water into lakes might cause substantial long-term problems. Thus, it involves special considerations. There are numerous examples from all over the world where disposal of agricultural drainage water into freshwater lakes, often in combination with reduced freshwater inflow, has added to the development of immense and catastrophic environmental problems. Examples include Lake Chapala in Mexico, Lake Manchhar in Pakistan, Lake Biwa in Japan, and other lakes where significant effort is required to halt environmental degradation.

DISPOSAL INTO EVAPORATION PONDS

In inland drainage basins without an adequate outlet to a river, lake or sea, or where disposal restrictions prevent discharge into rivers and lakes, one of the few options is disposal of drainage effluent into constructed evaporation ponds or natural depressions. Discharge into natural depressions has been practised for centuries. The impounded water is dissipated by evaporation, transpiration and seepage losses. Disposal into constructed ponds is practised worldwide. The planning and design of evaporation ponds have to take account of numerous environmental problems. These include:

- waterlogging and salinization problems in adjacent areas resulting from excessive seepage losses;
- salt-dust and sprays to areas downwind of the pond surface during dry and windy periods, which might damage vegetation and affect the health of humans and animals; and
- concentration of trace elements that might become toxic to fish and waterbirds because of bioaccumulation in the aquatic food chain.

The following examples from Pakistan, the United States of America and Australia highlight the opportunities and constraints of evaporation ponds.

Evaporation ponds in Pakistan

In Pakistan, evaporation ponds have been constructed under the Salinity Control and Reclamation Project (SCARP) VI for the disposal of the highly saline drainage effluent from the irrigated areas on the fringes of the desert located 500-800 km from the sea. SCARP VI has employed vertical drainage to lower and maintain the water table at a depth of 2.3 m over an area of 152 000 ha. The area is underlain by groundwater with an *EC* ranging from 15 to 34 dS/m. A total of 514 tubewells, ranging in capacity from 30 to 90 litres/s, have been installed to pump out 600 million m^3 annually. This water is discharged into open surface drains for conveyance to the evaporation ponds for final disposal. Allowing for conveyance losses, the drainable surplus to be disposed of is an estimated 540 million m^3. It was anticipated that the resultant salinity in the drainage water would be of the order of 30 dS/m (NESPAK-ILACO, 1981).

The design of the evaporation ponds assumed a percolation rate of 280 mm per year (half of the estimated value) and an evaporation rate of 1 700 mm per year (against the observed value of 1 800 mm). The surface area of the ponds was estimated to be 27 300 ha. The evaporation ponds were developed in a desert area on the fringes of the Indus Plain just beyond the canal command area of the project. The pond area is characterized by interdunal depressions (with highly sodic soils), between longitudinal sand dunes 4-9 m high. The underlying groundwater is highly brackish. For the development of the evaporation ponds, dykes were provided across the saddles and channels cut across the dunes to form a series of interconnected ponds. The ponds that have been developed so far have a surface area of 13 350 ha and a distance of 1-3 km separates their edges from the irrigated area. The drainage effluent is discharged into the evaporation ponds by gravity flow from one drainage system and pumped from another drainage system further downslope.

The operation of the ponds started in 1989 and by May 1998 it was reported that water had spread over a pond surface area of 4 200 ha (IWASRI, 1998). The period of operation has been too short to establish a state of equilibrium. However, soon after ponding started some irrigated areas close to the ponds were severely affected by waterlogging. This could be ascribed to significant seepage losses from the ponds generating a zone of high groundwater, obstructing or retarding the natural subsurface drainage from the irrigated lands (NRAP, 1991). The seepage affected area had grown to about 4 000 ha in 1993, even though the tubewell pumping was effective in lowering the water table to 3 m in parts of the drained area.

Evaporation ponds in California, the United States of America

Between 1972 and 1985, 28 evaporation ponds were constructed covering an area of about 2 800 ha receiving about 39 million m^3 annually of subsurface drainage from 22 700 ha of tile drained fields (SJVDIP, 1999d). Basinwide, the pond area is about 12 percent of the total area drained. Most of the ponds are located in the Tulare subarea, a closed basin in the San Joaquin Valley. The salt concentration in the waters discharged into these ponds ranged from 6 to 70 dS/m with an annual salt load of 0.88 million tonnes, about 25 percent of the annual salt load accumulating in the more than 0.9 million ha of cropland.

The concentration range of selenium, the principal constituent of concern, is from 1 to over 600 ppb. A selenium concentration of 2 ppb is considered the upper limit for aquatic life in ponds as selenium tends to bioaccumulate 1 000- to 2 500-fold in the aquatic food chain. Selenium is toxic to waterbirds as it substitutes for sulphur in essential amino acids resulting in embryo deformities and reduced reproduction rates. Ten ponds (with a surface area of about 4 900 ha) are active and managed by seven operators. The other ponds have been voluntarily deactivated due to the high costs of meeting the waste discharge requirements and mitigation measures for bird toxicity. The state/regional water quality regulatory agency ordered the closure of several ponds because of the unacceptable toxic effects of selenium on waterbirds from selenium present in the impounded drainage waters.

The waste discharge requirements for pond disposal define the requirements and compliance schedules designed to discourage wildlife use of evaporation ponds and/or provide mitigation and compensation measures to offset adverse effects on birds.

The evaporation ponds are constructed by excavating the soil to form berms with side-slopes of at least 3:1 (h:v). The pond bottoms are unlined but compacted, the pond environs are kept free of vegetation, and a minimum water depth is maintained at 60 cm. Migratory waterbirds attracted to the pond site are to be scared off, and bird diseases kept under control. Compliance monitoring includes seasonal drainage water and sediment selenium concentrations, and biological monitoring of birds for abundance and symptoms of toxicity. When adverse selenium impacts are noted, off-site mitigation measures must be implemented with either compensation or alternative habitats as guided by approved protocols and risk analysis methods. Compensation habitats are constructed to compensate for unavoidable migratory bird losses by providing a wetland habitat safe from selenium and predators. Alternative habitats are year-round freshwater habitats immediately adjacent to contaminated ponds in order to provide dietary dilution to selenium exposure. In spite of the stringent waste discharge requirements and associated high costs, farmers and districts in the Tulare subarea use evaporation ponds for drainage water disposal basins. This is because there are no opportunities for off-site discharge into the San Joaquin River or other designated sinks for drainage waters. Evaporation ponds are the only economic means of disposal of saline drainage in the Tulare-Kern subareas.

Evaporation ponds in Australia

The Salinity Drainage Strategy of the Murray-Darling Basin Commission imposes constraints on the amount of river disposal. Export of saline drainage water via pipelines to the sea has been considered but studies show that this option is relatively uneconomic compared to other disposal options (Leaney *et al.*, 2000). Saline disposal basins have been an important measure for the disposal of drainage effluent and will remain so. In the Murray-Darling Basin in 1995, 107 basins were actively used with a total area of more than 15 900 ha, a total storage capacity of more than 113 million m^3 and an annual disposal volume of more than 210 million m^3/year.

The use of regional-scale basins was a common approach in the Murray-Darling Basin. Regional basins were developed on the most convenient sites from an engineering standpoint. This sometimes had detrimental environmental, socio-economic and aesthetic impacts. These concerns together with the new opinion that beneficiaries of irrigation should be responsible for their own drainage management led to the development and use of local-scale basins. These basins can be in the form of on-farm basins which are privately owned and located on individual properties. They can also be privately or authority-owned community basins shared by a small group of properties. The choice between on-farm or community basins should consider physical, environmental and sociopolitical issues as well as costs. Economic analysis has shown that there is generally little difference in cost between the two options (Singh and Christen, 2000). Therefore, environmental and social considerations should outweigh the negligible economic differences in the choice between on-farm and community basins in the Murray-Darling Basin.

Design considerations for evaporation ponds

The first step in the design of an evaporation pond is to determine the volume of drainage water in need of disposal. The volume of drainage water depends on the land use, climate, irrigation practice and type of drainage system. The planning stage needs to include an assessment of the salinity of the drainage water as well as a full analysis for trace elements, nutrients and pesticides to prevent possible accumulation to toxic levels (Leaney *et al.*, 2000). Trace elements in drainage water disposed into the evaporation ponds include arsenic, boron, molybdenum, selenium, uranium and vanadium. The trace element of most concern in California for impounded drainage waters is selenium. The influent selenium is subjected to a number of reactions such as evapoconcentration, seepage losses, reduction of oxidized species of selenate (Se+6) and selenite (Se+4) to selenide (Se-2) as well as elemental selenium (Se0), methylation by microbes and plants and volatilization as dimethylselenide (DMSe), and adsorption of selenite to pond bottoms (Tanji and Gao, 1999). Approximately 60 percent of the mass of selenium influent appears to remain in the ponded water, the balance being immobilized or lost by the above mechanisms.

The hydrologic balance of evaporation ponds is relatively simple. The inputs into the pond include drainage water from cropped land, rainfall and, where applicable, water pumped into the pond from a perimeter drain installed to intercept seepage water from the pond. The outputs consist of evaporation, non-recovered seepage losses and transpiration from aquatic vegetation. An example of the hydrology of a typical pond system in California is: 1.91 m/year drain influent; 0.022 m/year precipitation; 1.87 m/year evaporation; 0.39 m/year seepage; and 0.46 m/year influent from the perimeter interceptor drain that collects both pond seepage losses and shallow groundwater.

The required pond area can be calculated on the basis of this hydrologic balance and depending on the depth of water layer that is to be maintained in the pond. When the salinity of the water in the evaporation ponds increases, the evaporation rate from the water surface decreases as a result of a decrease in water vapour pressure. The design of the pond area needs to take this aspect into account. Evaporation rates in ponds have been measured. The pond evaporation can be estimated using an empirical correction factor (*Y*) and *ETo* (Johnston *et al*., 1997):

$$E = Y(ET_O) \tag{21}$$

in which $Y = 1.3234 - 0.0066\ EC$ for water between *EC* 14 and 60 dS/m.

Seepage losses from evaporation ponds depend on soil, geohydrological and topographical conditions. Site selection is very important for preventing excess seepage losses. Depending on the hazards that excessive seepage might occur and depending on the environmental regulations, lining and a seepage collection system may be required in order to ensure that there will be no contamination of groundwater, and to prevent waterlogging in adjacent areas. The availability of materials should determine the type of lining. Research from Australia has shown that a small amount of controlled seepage of around 0.5-1.0 mm/d is required to maintain evaporative disposal capacity, as evaporation rates decrease substantially under hypersaline conditions (Leaney *et al.*, 2000). This implies that basins should be located preferably within the drained area. Where the basin is sited outside the limits of the drainage system, it should be located within a specific salt containment area equipped with effective interception and recycling works.

For those pond systems deemed to be hazardous to waterbirds, pond operators are required to establish alternative and compensation habitats. Alternative habitats using freshwater are built within the functional landscape of evaporation ponds to provide year-round habitat to dilute the diet to the feeding birds. Compensation habitats are located outside the functional landscape of the evaporation ponds (about 5 km) during the breeding season.

Injection into deep aquifers

Disposal by injection into deep aquifers is a process in which drainage water is injected into a well for placement into a porous geologic formation below the soil surface (Lee, 1994). The oil and gas industries use deep-well injection to dispose of waste brine. In California, the United States of America, deep-well injection technology has been used for over 60 years for the disposal of oil field brines. The technology could potentially be applied for the disposal of agricultural drainage water provided the conditions of the receiving geologic formation are adequate, there are no environmental hazards and the costs are not prohibitive. The main environmental concern is leakage from pipes and confining aquifers. Drainage effluent can contaminate freshwater zones through leakage.

The drainage and aquifer water should be compatible, as the mixture should not produce precipitates, which clog the wells. Furthermore, the receiving geologic formation should have sufficient porosity and thickness to receive the injected water. If water containing nitrate is injected into aquifers containing organic matter and ferrous iron, the growth of nitrate reducing bacteria might clog the pores of the receiving formation as they accumulate (Westcot, 1997).

The Westlands Water District, California, the United States of America, carried out a prototype deep-well injection to dispose of up to 4 000 m^3/d of drainage water (Johnston *et al*., 1997). The drainage water was to be in injected into shale and sand formations 1 554 and 2 164 m, respectively, beneath the ground surface. The well was drilled to a total depth of 2 469 m at a cost of about US$1 million. The casing was perforated with 13 perforations per metre from depths of 2 245 to 2 344 m and from 2 411 to 2 414 m in the Martinez geologic formation for a total length of perforations of 102 m. Following the recovery of some natural formation water samples, an injection test was conducted. The fluid injected was irrigation water filtered through a 0.5-micron filter and treated with 2-percent potassium chloride and a chlorine biocide. The filtered and treated water was injected through a 4.75-cm tubing at a rate of 12 litres/s for a total of 175 000 litres of water injected. Then, a 48-hour pressure fall-off test was conducted that revealed the permeability of the geologic formation was 12 mm/d. This permeability value is too low to achieve the desired injection rate of 44 litre/s. In spite of the filtration and chlorination of the injected water, plugging of conducting pores occurred in the shale. The United States Environmental Protection Agency rejected a request for permission to conduct a second injection test in the sandy, more permeable formation above the shale. The Westlands Water District abandoned this method of disposal.

Chapter 8
Treatment of drainage effluent

Need for drainage water treatment

Treating drainage water is normally one of the last drainage water management options to be considered. This is due to the high costs involved and to uncertainty about the treatment level achievable. The treatment of drainage water should be considered where all other drainage water management measures fail to guarantee safe disposal or where it is financially attractive. For subsurface drainage water containing very high levels of salinity, selenium and other trace elements, the treatment objectives are: i) reduce salts and toxic constituents below hazardous levels; ii) meet agricultural water management goals; iii) meet water quality objectives in surface waters; and iv) reduce constituent levels below risk levels for wildlife.

Treatment options

The treatment of agricultural drainage water presents a challenge due to the complex chemical characteristics of most drainage waters (Lee, 1994). Table 23 details the average chemical quality of subsurface drainage waters disposed into Kesterson Reservoir in the San Joaquin Valley as well as those disposed into evaporation ponds. The drainage waters are saline and of the $NaCl$-Na_2SO_4-type water. The waters conveyed by the San Luis Drain came from a single site in Westlands Water District in contrast to the evaporation pond waters that came from 27 sites.

Table 23
Average composition of agricultural tile drainage water in the San Luis Drain (drainage waters disposed into evaporation basins in parenthesis)

Constituent	Concentration ppm	Constituent	Concentration ppb
Sodium	2 230	Boron	14 400 (25 000)
Calcium	554	Selenium	325 (16)
Magnesium	270	Arsenic	1 (101)
Potassium	6	Molybdenum	ND (2 817)
Alkalinity as $CaCO_3$	196	Uranium	ND (308)
Sulphate	4 730	Vanadium	ND (22)
Chloride	1 480	Strontium	6 400
Nitrate	48	Total chromium	19
Silica	37	Cadmium	<1
TDS	9 820 (31 000)	Copper	4
Suspended solids	11	Lead	3
Total organic carbon	10.2	Manganese	25
COD	32	Iron	110
BOD	3.2	Mercury	<0.1
		Nickel	14
		Zinc	33

Source: SJVDP, 1990; and Chilcott *et al.*, 1993.

There are numerous wastewater treatment processes for industrial and urban wastewater and for the preparation of drinking-water. Many of them offer potential for the treatment of agricultural drainage water. Treatment processes for drainage water can be divided into processes that reduce the total salinity of the drainage water and processes that remove specific ions. Methods for the removal of trace elements can be biological, physical and chemical.

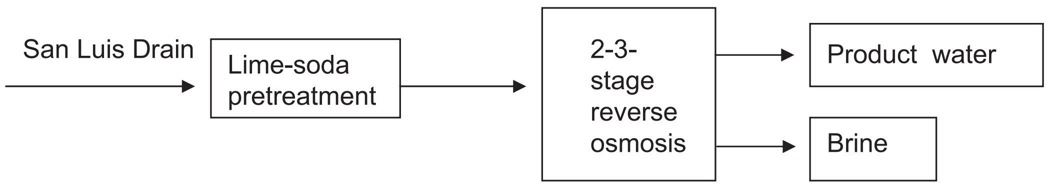
Most desalinization processes also remove trace elements but their costs are often prohibitive. Less costly methods for the removal of trace elements are being developed. Lee (1994) has reviewed treatment technologies for drainage water. The SJVDIP (1999b) has reviewed treatment technologies for removing selenium from agricultural drainage water. The following is a brief summary of their findings.

Desalinization

There are numerous desalinization processes including ion exchange, distillation, electrodialysis and reverse osmosis. Of these processes, reverse osmosis is considered to be the most promising for the treatment of agricultural drainage water mainly due to its comparatively low cost.

Reverse osmosis is a process capable of removing different contaminants including dissolved salts and organics. In reverse osmosis, a semi-permeable membrane separates water from dissolved salts and other suspended solids. Pressure is applied to the feed-water, forcing the water through the membrane leaving behind salts and suspended materials in a brine stream. The energy consumption of the process depends on the salt concentration of the feed-water and the salt concentration of the effluent. Depending on the quality of the water to be treated, pretreatment might be crucial to preventing fouling of the membrane. Figure 44 describes one of several pretreatment reverse osmosis systems studied in the San Joaquin Valley. Other pretreatment steps could be lime treatment along with ion exchange.

Figure 44
Reverse osmosis system with lime-soda pretreatment



Source: after CH2M HILL, 1986.

Table 24 presents the results of a trial-run reverse osmosis using the lime-soda softening pretreatment (CH2M HILL, 1986). The permeate is the product (desalted) water and the concentrate is the brine water. The results show that *TDS* can be desalted from 9 800 to 640 ppm, boron from 14.5 to 7.6 ppm, and selenium from 325 to 3 ppb in a three-stage reverse osmosis system. The efficiency of removal declines with stages.

The California Department of Water Resources conducted pilot-plant-scale reverse osmosis of saline drainage using cellulose acetate membranes. The bacterial and chemical fouling of the membrane was a major problem. The drainage water had to be treated with alum, and passed through a sedimentation pond and a chlorinated and filtration system. In spite of this level of pretreatment, the membranes tended to foul due to the precipitation of gypsum and calcite. The drainage waters are saturated with respect to calcite and gypsum. This same chemical fouling

TABLE 24
Results of a trial-run for a three-stage reverse osmosis system, lime-soda pretreatment

Description	*TDS* ppm	Sodium ppm	Chloride/nitrate ppm	Sulphate ppm	Boron ppm	Selenium ppb
Influent	9 793	2 919	1 550	5 010	14.5	325
Stage 1 concentrate	19 346	5 721	3 038	9 970	23.4	650
Stage 1 permeate	240	117	62	50	5.4	0
Stage 2 concentrate	38 071	13 156	5 924	19 791	38	1 298
Stage 2 permeate	614	286	152	150	8.8	1
Stage 3 concentrate	73 022	22 107	15 987	38 650	62	2 579
Stage 3 permeate	1 480	669	355	396	14.3	3
Overall permeate	640	176	155	201	7.6	3

Source: CH2M HILL, 1986.

problem is being faced by the Yuma desalting plant off the Colorado River using drainage waters from the Wellton-Mohawk irrigation project. The estimated cost of desalting is more than US$0.81/$m^3$, too expensive for irrigated agriculture but possibly affordable for municipalities with freshwater shortages. This cost does not include the management and disposal of the brine water. However, a potential exists for partially treating the average 10-dS/m-drainage water to about 2-3 dS/m for use by agriculture and wildlife.

Trace element treatment

As the technology of reverse osmosis is experimental and expensive, cheaper methods of removing toxic trace elements are being pursued.

Biological processes

Conventional column reactor systems have been utilized to remove selenium from drainage waters (SJVDIP, 1999b). Selenium is microbially reduced to elemental selenium under anoxic (anaerobic) conditions in the presence of organic carbon sources (Owens, 1998).

$$\underset{\text{(soluble selenate)}}{\text{Se(+6)} + \text{bacteria} + \text{organic carbon}} \longrightarrow \underset{\text{(soluble selenite)}}{\text{Se(+4)}} \longrightarrow \underset{\text{(elemental selenium particulates)}}{\text{Se(0)}}$$

In the initial study, the biological reactor consisted of a two-stage upflow anaerobic sludge blanket reactor followed by a fluidized bed reactor. As selenium cannot be reduced while nitrates are present, a key treatment process is the reduction of nitrates prior to enhancing selenium reduction. The sludge blanket was seeded with inoculum from sludges from ordinary sewage treatment plants. This system yielded 30 ppb selenium product water.

A subsequent large-scale pilot study examined seven different reactor systems after upflow through a conical bottom liquid-gas-solid separator with the addition of methanol as the carbon source. The conical separator was seeded with granular sludge from a bread-making bakery. This first step reduced the average nitrate concentration from 45 to 3 ppm. The waters were then fed to a number of packed bed column reactors. The best sustained results were about a 90-percent removal of selenium from 500 to 50 ppb.

Biological treatment normally refers to the use of bacteria in engineered column reactor systems for the removal or transformation of certain constituents, e.g. organic compounds, trace elements and nutrients (Owens and Ochs, 1997). However, biological treatment also includes algal-bacterial treatment processes and wetland systems. Much research has focused on the removal of selenium from drainage effluent. Box 12 describes an example of the basics of an algal-bacterial system for the removal of selenium (SJVDIP, 1999b).

Box 12: Basics of an algal-bacterial system for the removal of selenium

The concept of the algal-bacterial selenium-removal process is to grow micro-algae in the drainage water at the expense of nitrate and then to utilize the naturally settled algal biomass as a carbon source for native bacteria. In the absence of oxygen, the bacteria reduce the remaining nitrate to nitrogen gas and further reduce selenate to insoluble selenium. The insoluble selenium is then removed from the water by sedimentation in deep ponds and, as needed, by dissolved air flotation and sand filtration. Supplemental carbon sources such as molasses can be employed as reductant in addition to algal biomass. A prototype algal-bacterial selenium-removal system reduced the selenium content in water from 367 ppb (influent) selenium to 20 ppb (effluent).

Chemical processes

Chemical treatment processes refer to the use of chemicals to remove trace elements from polluted wastewater. Chemicals are frequently used for industrial wastewater treatment but are not effective in agricultural drainage water due to their often complex chemical characteristics (Lee, 1994). Chemical processes have been developed for the reduction of selenate to elemental selenium by means of ferrous hydroxide. Under laboratory conditions, ferrous hydroxide was able to reduce and precipitate selenium by 99 percent in 30 min. In field studies, although 90 percent of the selenate was reduced, the reactor time required was up to 6 h. It appeared that dissolved bicarbonate, oxygen and nitrate influenced the reduction process.

Physical processes

Physical processes involve the adsorption of ions on natural and synthetic surfaces of active materials, including ion exchange resins. Box 13 provides an example of a mini-pilot plant for the removal of heavy metals.

Box 13: Mini-pilot plant for the removal of heavy metals

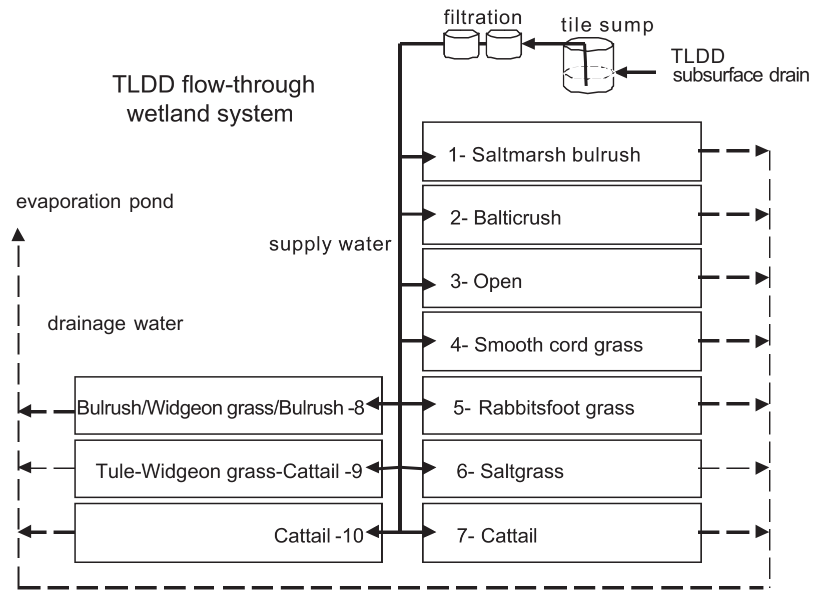
Harza Engineering Co. tested a pilot-scale treatment plant in 1985. The processes used iron filings in flow-through beds. The principle was based on the idea that oxygen could activate the surface of the iron, which could then adsorb selenium. The testing was discontinued as the beds quickly cemented with precipitates. The advantage of zero-valent iron is that it can reduce the concentration of selenium to very low concentrations. This method could be used as a polishing step following microbial treatments. Where the waste is anaerobic after microbial treatment, the formation of secondary precipitates is minimized.

Flow-through artificial wetlands

Figure 45 shows the layout of a pilot project for removing selenium by flow-through wetland cells conducted in the Tulare Lake bed, a closed basin of the San Joaquin Valley (Tanji and Gao, 1999). The goal was to remove selenium from drainage waters to a bird-safe level prior to disposal into evaporation ponds.

Tile drainage effluent containing about 20-ppb selenium from an adjacent farm was passed through a sand bed filter system and metered into the cells (15.2 x 76 m) with a variety of substrates (vegetation). The inflow water was measured twice a week by a totalizing meter. The water depth in Cells 1-7 was maintained at about 20 cm, and outflow was measured by v-

FIGURE 45
Layout of pilot-scale constructed wetland experimental plots at the Tulare Lake Drainage District



Source: Tanji and Gao, 1999.

notch weir. Cells 8-10 had variable water depths of about 20 cm, 60 cm where widgeon grass (*Ruppia*) was grown. The target residence time for the flowing waters was 7 days for Cells 1-7, 21 days in Cells 8 and 9, and 14 days in Cell 10. These residence times were selected after preliminary runs for optimal removal. A residence time of three days was too short for selenium removal and a residence time of more than 21 days did not increase selenium removal. Seepage rates in the cells were about 1 cm/d and evapotranspiration slightly greater than *ETo* (annual value about 1 600 mm).

Table 25 presents the performance results for the year 1999 with average weekly water selenium of 18.2 ppb, over 90 percent in the selenate form (Se+6). The residence times achieved were reasonably close to target values considering the variability in monthly *ETo*. The selenium concentration in the outflow waters varied from 4.6 to 12.3 ppb. The ratio of outflow to inflow

TABLE 25
Performance of the wetland cells in removing selenium from drainage water with 18.2-ppb selenium

Wetland cell	Residence time days	Outflow selenium ppb	Outflow/inflow selenium conc. ratio	Outflow/inflow selenium mass ratio
1-Saltmarsh bulrush	10.3	6.1	0.33	0.07
2-Baltic rush	7.4	8.6	0.45	0.54
3-Open	7.5	12.3	0.68	0.57
4-Smooth cordgrass	9.7	6.7	0.37	0.24
5-Rabbitsfoot grass	8.4	10.3	0.55	0.11
6-Saltgrass	9.2	4.6	0.25	0.03
7-Cattail, shallow	7.0	11.6	0.63	0.59
8-Bulrush/Ruppia/Bulrush	24.1	10.5	0.57	0.21
9-Tule/Ruppia/Cattail	22.3	9.6	0.53	0.30
10-Cattail, deep	17.9	6.4	0.35	0.21

Source: Tanji and Gao, 1999.

selenium concentration ranged from 0.25 to 0.68 (a small ratio indicates high selenium removal). The ratio of outflow to inflow selenium on a mass basis ranged from 0.07 to 0.57 or 93 to 43 removal. The cell with open water had reduced selenium because algae and microbes naturally populated the cell and contributed to some selenium removal. In terms of performance, the ratio based on mass of selenium is a good indicator. However, in terms of potential impact on birds, the outflow concentration and ratio based on concentration are better indicators.

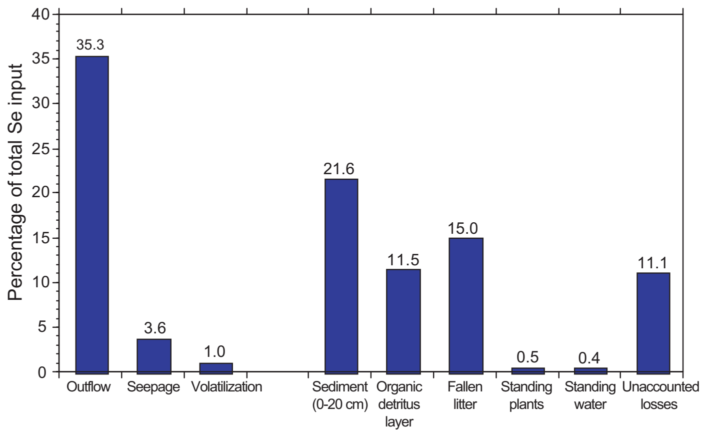
The control volume for each cell is the standing water, plants and the rootzone. Thus, the mass flux balance on selenium for each cell is:

$$\frac{\Delta M_{Se}}{\Delta t} = M_{Se,\,inflow} - M_{Se,\,outflow} - M_{Se,\,seepage} - M_{Se,\,volatilization} \qquad (22)$$

The righthand-side terms of Equation 22 are mass fluxes, and mass (M_{Se}) is defined as the product of selenium concentration and water volume, except for the volatilization term. Water inflow and outflow was monitored twice a week, water seepage estimated from the difference from inflow and outflow and ET_{crop} from $ETo * Kc$, where Kc is the crop coefficient. Volatilization of selenium by microbes and plants was monitored monthly. The $\Delta M/\Delta t$ is the mass flux of selenium accumulating in the control volume (cell) consisting of the sediments, organic detrital matter, fallen litter, standing water and standing crop.

Figure 46 presents a summary of the mass balance on selenium in the ten wetlands cells from July 1997 to September 2000. The values reported are based on the percentage of the mass of selenium in the inflow water. On average, about 35 percent of the mass inflow of selenium remained in the treated outflow water, with smaller percentages lost through seepage and volatilization losses.

FIGURE 46
Initial estimate on mass balance of selenium in ten flow-through wetland cells, 1997-2000



The remainder of the selenium accumulated in the cell as selenium present in the sediments, organic detrital matter, fallen litter, standing water and standing plants. The values reported are the mass of selenium found in the cells in September 2000. About 11 percent of the total selenium could not be accounted for due to errors in sampling and monitoring over a four-year period, and the difficulties of analysing for reduced forms of selenium. The sink mechanisms removing selenium from the floodwater were: adsorption of selenite (Se+4) to the mineral sediments mainly in the top 10 cm or so; selenium immobilized into elemental selenium (Se0) due to reduced conditions in the organic detrital layer; and organic forms of selenium (Se-2) tied up with the detritus and fallen litter. The principal removal mechanisms were adsorption and immobilization into elemental selenium and organic selenium.

The recommended selenium water standard to protect waterbirds is 2 ppb. None of these cells achieved that level of remediation but many cells certainly will reduce selenium toxicity.

However, outflow waters from these cells contain organic selenium (17-33 percent of the total selenium), which is more toxic than inorganic forms to wildlife. These and other results are currently being reviewed to determine whether selenium removal flow-through wetland cells is a viable treatment option

EVALUATION AND SELECTION OF TREATMENT OPTIONS

The first steps in the selection of any drainage treatment process are: i) define the problem; ii) determine the reasons for the required treatment; and iii) determine what is to be achieved. The main reason for opting for drainage water treatment is normally the desire to reuse the drainage effluent or to conform to regulatory disposal requirements. For both purposes, specific water quality criteria apply.

In order to make a preliminary selection of suitable treatment processes, it is necessary that sufficient data be available. These data consist of historical data on the chemical constituents of the drainage water, seasonal flow variations and variations in the concentrations of the constituents of concern. Once combined with information on the targeted quality of the treated effluent, it is then possible to shortlist drainage treatment processes that are theoretically suitable.

The technical capability of the treatment process is an important factor in the selection of a treatment technology. However, it is important to consider economic, financial, social and institutional criteria in order to ensure the sustainability of the treatment facilities.

Part II

Summaries of case studies from Central Asia, Egypt, India, Pakistan and the United States of America

Summaries of case studies

DRAINAGE WATER MANAGEMENT IN THE ARAL SEA BASIN

By *V. Dukhovny, K. Yakubov, A. Usmano and M. Yakubov*

Summary

The Aral Sea Basin is located in Central Asia and covers an area of 154.9 million ha. Under farming conditions characterized by high evaporation and low precipitation, most crops are irrigated. The irrigated area covered 7.9 million ha in 1999. The main crops grown under irrigation are cotton, rice, wheat, maize and fodder crops.

Water resources in the region consist of renewable surface and groundwater as well as return flows in the form of agricultural drainage water and wastewater. The two major rivers in the basin are the Amu Darya River in the south and Syr Darya River in the north. The total mean annual river runoff is 116 000 million m^3. The annual amount of groundwater that can be subtracted without damage is 13 100 million m^3. Water use in the Aral Sea Basin ranges between 110 000 million and 117 000 million m^3 annually depending on the actual water availability. The groundwater abstracted is about 10 000 million m^3 per year. On average, the agriculture sector accounts for 95 percent of all water withdrawals.

Each year about 5 000 million m^3 of water reaches the Aral Sea, compared with 54 000 million m^3 before the large-scale expansion of irrigation. This has led to the gradual drying up of the Aral Sea, which has had severe adverse effects on the region's environment, health and economy. It has been estimated that at least 73 000 million m^3 of water would have to be discharged to the Aral Sea each year for a period of at least 20 years in order to restore it to its 1960 level of 53 m above sea level. The riparian states consider this target to be unrealistic. Proposals have been made to restore part of the Aral Sea to a level of 38-40 m above sea level, requiring an inflow of at least 6 000-8 000 million m^3.

Low irrigation efficiencies have resulted in rising groundwater levels and secondary soil salinization. Nearly half of the irrigated lands are affected by salinity. Salt balances show that especially in dry years in the middle and lower reaches of the Amu Darya River salt accumulation takes place amounting to 0.6-10 tonnes/ha annually. In the lower reaches, even in wet years salt accumulation amounts to 8 tonnes/ha annually. For the Syr Darya Basin this phenomena can be observed in its middle reaches with an annual salt accumulation of 5.3 tonnes/ha. In addition to salinity, on 30 percent of the irrigated lands shallow groundwater poses a major problem for agricultural production. To combat these problems, a collector drainage network has been developed on 4.45 million ha. Within this area, 1.8 million ha have been equipped with horizontal or vertical subsurface drains. The disposal of drainage water causes considerable problems in terms of downstream water quality. About 92 percent of the total return flow consists of agricultural drainage water. Most of this water, about 20 000 million m^3/year, returns into the river systems

and a slightly smaller part, about 15 million m^3/year, is diverted to desert sinks. In total, 137 million tonnes of salt are discharged annually together with agricultural drainage water. Of this total, 81 million tonnes were originally present in the irrigation water and 56 million tonnes originate from the mobilization of salts from the subsoil.

The guiding principle in the planning and management of water resources in the Aral Sea Basin is to set targets for water use and conservation, and to establish mechanisms to reach these targets. A wide range of measures has been implemented to decrease the losses from irrigated areas (including replacing cotton with wheat, which requires less water). Extensive research has been undertaken to establish practices that reduce application, conveyance and operational losses. Research has shown that application efficiencies can be improved by furrow levelling and rescheduling the amount and timing of irrigation water applications. One of the main reasons for the present waterlogging and salinity problems is the high irrigation norms adopted in the past. Applying a larger number of irrigation turns with smaller amounts of water increases the efficiency. However, research results have not led to large changes in irrigation practices. The large-scale adoption of research results requires an effective extension service. Another obstacle is that the infrastructure and irrigation scheduling on former collective farms do not always allow individual farmers to apply the amounts of water that would be most efficient. In addition to on-farm losses, high distribution losses are a major concern. Following the breakup of the former collective farms, new allocation and scheduling rules have yet to come into existence. The establishing of water users associations could help reduce distribution losses within the former collective farms. However, such reductions can only be expected if irrigation supply from higher levels also improves.

Drainage water use for irrigated agriculture in its place of origin or in adjacent areas is one of the options for reducing the disposal problems in the Aral Sea Basin. For the Central Asian conditions, a water quality classification has been developed for the use of saline water and conditions for use in relation to its salinity and chemical composition. A soil classification for the use of saline water has also been proposed. It is estimated that drainage water use can be increased to up to 25 percent of the annual drainage flow in the Aral Sea Basin, compared with the current 11 percent. In addition, agricultural drainage water can be reused in wetlands and biodiversity development.

Along the Amu Darya River, desert depressions are found in which drainage effluent is disposed and left for evaporation. These desert sinks or drainage lakes are of various sizes. This practice is a feasible alternative to disposal into the river. Moreover, it avoids an increase in mineralization of the river. Specific forms of flora and fauna have become established around these lakes, and fisheries may become a possibility. The main problem is that most lakes along the Amu Darya River have reached their maximum capacity. If the inflow into the lakes is not controlled, there is a risk that the lakes will overflow and flood the surrounding areas.

There have been plans to construct an outfall drain to the Aral Sea since the 1980s. This drain would run parallel to the Amu Darya River for 900 km. Upon its completion, mineralization levels in the river could be maintained below 1 g/litre at all times. The drain would collect drainage water from an area exceeding 1 million ha. Construction work commenced in the early 1990s. However, due to a lack of funds, work stopped in 1994 with only 20 percent of the drain completed.

DRAINAGE WATER REUSE AND DISPOSAL: A CASE STUDY FROM THE NILE DELTA, EGYPT

By *N.C. Kielen*

Summary

The Nile Delta in northern Egypt starts north of Cairo where the Nile River splits into two branches, the Domitta and Rozetta. Egypt's cultivated land amounted to 3.3 million ha in 1994, nearly all of it irrigated. In the absence of effective rainfall, the country's water resources consist of: surface water from the Nile River; shallow and deep groundwater; and drainage water. The main source of water in Egypt is the Nile River. The 1959 agreement with Sudan allocates 55 500 million m^3 of the Nile discharge per year to Egypt. The annual recharge to the groundwater in the Nile Valley and Nile Delta is estimated to be between 5 600 million m^3 and 6 300 million m^3.

The gross water use in the mid-1990s was about 60 300 million m^3 per year, of which 51 500 million m^3 or 85 percent was extracted for irrigation purposes. The Ministry of Water Resources and Irrigation (MWRI) expects irrigation demand to increase to 61 500 million m^3 per year by 2025. The projected total water demand cannot be met by developing new water resources. Besides increasing water use efficiency, drainage water reuse is the most promising immediate and economically attractive option to make more water available for agriculture. In the 1980s, the reuse of agricultural drainage water became a policy to augment Egypt's limited fixed freshwater resources and to close the gap between supply and demand.

Reuse is centrally organized with the pumping of water from the main drains into the main canals. In 1996/97, the amount of water pumped at the reuse mixing stations was 4 400 million m^3 with an average salinity of 1.8 dS/m. The total quantity of drainage water released to the Mediterranean Sea and coastal lakes was 12 400 million m^3 with an average salinity of 4.2 dS/m. The MWRI has committed another 3 000 million m^3 of the drainage water for reuse within the new reclamation areas of the El Salam Canal and Umoum Drain projects. Another 1 000 million m^3 of the drainage water will be reused in the near future to irrigate newly reclaimed lands in the Kalapsho area in the Middle Delta. Therefore, the volume of drainage water officially reused for irrigation is expected to increase to 8 000 million m^3 per year in the near future.

Farmers also use drainage water directly by pumping it from drains close to their fields. This is termed unofficial reuse. Estimates of the amount of drainage water unofficially used for irrigation range from 2 800 million m^3 to 4 000 million m^3 per year. Unofficial reuse in illegal rice fields in the Bahr Hadus area is of concern to the MWRI as it competes with the El Salam Canal drainage diversion.

A central issue in water resource management in Egypt is how much of the annual drainage discharge can be reused. In theory, 13 300 million m^3 of drainage water with a salinity of less than 4.7 dS/m is available for reuse. This amount is equivalent to 84 percent of the generated drainage in 1993/94. Taking leaching requirements and deteriorating drainage water quality due to municipal and industrial pollution into consideration, the total estimated reuse potential is 9 700 million m^3 with a maximum salinity of 3.5 dS/m, of which 8 000 million m^3 can be used effectively. This is 5 000 million m^3 more than the reuse level in 1993/94.

The Northern Lakes located adjacent to the Mediterranean Sea, comprising lakes Maruit, Edko, Burrulus and Manzala, are economically important as they support a large fishery and many fish farms. For continued fish production, the salinity levels in the lakes should be maintained between 5.5 and 6.25 dS/m. Based on maintaining these salinity levels, the drainage outflow to

Lake Manzala and Lake Edko can be reduced by a maximum of 50 percent of the outflow level in 1993/94. The outflow to Lake Burrulus is already on the low side and cannot be reduced any further. The additional drainage reuse potential based on sustained freshwater lake fisheries is 4 000 million m^3 per year, which is 1 000 million m^3 less than the reuse potential based on maintaining a favourable salt balance in the Nile Delta.

The official strategy for drainage water reuse has not caused major increase in soil salinity levels on a large scale. In terms of maintaining a favourable salt balance in the Nile Delta, drainage reuse in Egypt has been successful. Factors that have contributed to a sustainable implementation of reuse are that drainage has been implemented on 90 percent of the irrigated lands, and that reused drainage water after mixing with freshwater has a low salinity content. Salinity levels of the water in the main drains increase from south to north, which is the general flow direction. Therefore, drainage water with favourable salinity levels is intercepted for reuse while drainage water with a high salinity content is disposed in the coastal lakes and Mediterranean Sea. However, soil salinity levels might be high locally especially in tail end areas where irrigation water is inadequate and groundwater salinity is high.

Since the 1990s, pollution of the drains as a result of large-scale urbanization and industrialization has received increased attention. Due to the increasing deterioration of water quality in the main drains and the increasing concern about how to manage unofficial reuse of drainage water, it appeared that it would be difficult to expand official reuse. The MWRI explored new opportunities for drainage water reuse. Between the centralized official reuse and the localized unofficial reuse, there is the option of capturing drainage water from branch drains and pumping it into the branch canals at their intersections. This level of reuse is termed intermediate reuse. It offers two main advantages. First, relatively good-quality drainage water is captured before discharging into the main drains where it is lost to pollution. Second, with the poor level of the current delivery system, water shortages often occur at the tail end of canals. At intermediate level, drainage water is pumped into the tail end of the branch canals, so making water available to water shortage areas.

Two alternative modes of reuse have been tested on an experimental basis. The first involved the application of freshwater separately from drainage water in the so-called cyclic mode. The other option tested was deficit irrigation in which irrigation was withheld during a certain period. In this period, the crops took up shallow groundwater to satisfy their water requirements. This strategy seemed to offer considerable scope in water shortage areas. Additional reported advantages are: it saves on engineering and energy costs for pumping; it avoids farmers having to come into contact with contaminated water; it prevents the application of saline water to the upper rootzone layers; and it reduces nitrate pollution of drainage effluent. Leaching should be applied periodically to guarantee long-term sustainable salinity levels. The alternative drainage reuse strategies tested yielded reasonable results in terms of soil salinity and crop yields.

Drainage water reuse and disposal in northwest India

By *N.K. Tyagi*

Summary

Northwest India encompasses two major river basins (the Indus and the Ganges) and lies in the states of Punjab (south), Haryana and Rajasthan (northeast and northwest). Although it is a water deficient region, the introduction of canal irrigation has reduced the gap between supply

and potential demand to a certain extent. Irrigation is the mainstay of agriculture in this area. As irrigation development took place without the parallel development of drainage, water and salt accumulation has occurred in most canal command areas. Salinity has already affected an area of 1 million ha. This area might expand to more than 3 million ha in the next 20 to 30 years unless remedial measures are taken.

In Punjab and Haryana, surface drains were constructed and groundwater development and flood control were initiated in order to overcome waterlogging and salinity problems. In areas where surface drains do not have a natural drainage outlet, low-head high-discharge pumps dispose the drainage effluent into major canal systems. Horizontal subsurface drainage has been developed on a small scale in the region. Vertical drainage in the form of shallow tubewells is widespread throughout the region. The density of the tubewells varies with the groundwater quality and recharge. In areas located in the saline groundwater zones, the annual groundwater recharge from the extensive canal network continues to exceed groundwater abstraction. As a result, the rise in the water table and subsequent salinization has continued in these areas. If national food security is to be ensured, this problem needs to be addressed. Efforts to date have consisted of lining irrigation channels up to watercourse level and sinking public tubewells for irrigation. Water testing facilitates have been improved and the technology for the use of tubewell water has been disseminated to farmers.

The reuse of saline effluents is an important option for Northwest India as it could supplement scarce irrigation water supplies and also help to alleviate disposal problems. Reuse can take place by applying the drainage water directly to the crop, blending it with canal water and using it intermittently with canal water. The last form of reuse is most common for private tubewell water use as farmers receive canal water for only a few hours per week. The scope for reuse is highest during the winter season when evaporative demands are low and the initial soil salinity is low due to the leaching which occurs during the monsoon rains between June and September. During the winter season, the soil salinity will increase slowly. When summer starts, the crops are at maturity stage and are able to tolerate higher levels of salinity. The subsequent monsoon rains will leach the salts that have accumulated during the winter and early summer. If the monsoon rains are not sufficient, a heavy pre-irrigation might be given to ensure that the salinity is reduced to acceptable levels for a good germination of the subsequent winter crop.

In addition to drainage water reuse, shallow water table management is an important mechanism for the use of soil water in lands provided with drainage. Experiments showed that on a sandy loam soil with a water table at a depth of 1.7 m and with a salinity of 8.7 dS/m, the water table contributed up to 50 percent of the water requirement when irrigation was withheld. Similarly, at another site, a shallow water table at a depth of about 1.0 m and with a salinity of 3.0-5.5 dS/m facilitated the achievement of potential yields even when the surface water application was reduced to 50 percent. The salinity buildup was negligible and the small amount of accumulated salts was leached in the subsequent monsoon season.

For the maintenance of a favourable salt balance in soil and groundwater, the salt outflow from the system should at least equal the salt inflow plus any salt generation in the system itself. For Northwest India, the possible methods of disposing of the drainage effluents include: (i) disposal into the regional surface drainage system that links the major rivers flowing through the region; (ii) pumping into the main and branch canals that carry high flow discharges for most of the year; and (iii) disposal into evaporation ponds.

During the monsoon period, the Yamuna River carries a very high flow discharge. The salinity of the river water during the monsoon is less than 0.2 dS/m. The high flow and low salinity of the water during the monsoon period indicate the potential for the disposal of saline effluents into this river. In Haryana, an area of 1 633 000 ha drains into the Yamuna River. An extensive surface drainage system to evacuate floodwaters has been constructed. Projections regarding the amount of subsurface drainage water that could be disposed through the surface drains into the Yamuna River were made on the basis of a study covering a period of five years. In the basin, waterlogging and salinity affect 183 000 ha. The study shows that the entire amount of subsurface drainage water could be disposed into the Yamuna River during the monsoon period, whereby the river water salinity would remain below 0.75 dS/m. In the winter months, low river flows reduce the disposal capacity considerably. However, in this period, the disposal requirements can be reduced through the implementation of shallow water table management and the reuse of generated drainage water.

As the need for saline water disposal is increasing, evaporation ponds might need to be constructed. The performance of an evaporation pond constructed in a sandy area at Hisar was not encouraging, probably the result of locating the pond in a slightly higher area. It may be necessary to construct a series of ponds in the lowest-lying areas.

It will be necessary to maintain a fine balance between reuse and disposal in order to establish a favourable salt regime in the region. The existing experience from small pilot projects is only indicative of the feasibility of reuse, shallow water table management and disposal requirements. The large-scale drainage disposal and reuse programmes planned for Northwest India will need to give due weight to maintaining a favourable salt regime at basin level.

Drainage water reuse and disposal: a case study on Pakistan

By *M. Badruddin*

Summary

The Indus River and its tributaries are the main sources of water in Pakistan. Under the Indus Water Treaty of 1960 between India and Pakistan, Pakistan is entitled to all the waters of the Indus and to that of two out of the five eastern tributary rivers, the Jehlum and Chenab. Since then the annual inflow has averaged 171 460 million m^3. The quality of the water in the Indus River and its tributaries at their entry points into Pakistan is characterized by a low salt content ranging from 0.16 to 0.47 dS/m. Apart from surface water, groundwater is an important source of supplemental irrigation supplies in the irrigation system of Pakistan. Estimates of the early 1990s indicate that 44 520 million m^3 is pumped annually within the canal commands both from private and public tubewells. The salinity content of groundwater varies considerably. About one-third of the irrigated area has groundwater with a high salt content.

Irrigation in Pakistan is based on the water supplies of the Indus River and its tributaries and is essentially confined to the Indus Plain. The surface irrigation system consists of the command areas of 43 main canals and covers the largest contiguous irrigated area in the world extending over a gross command area of 15.8 million ha. Due to the absence of any entrenched waterways in the flat plains (which could provide natural drainage), the introduction of irrigation caused a gradual rise in the water table. This has resulted in widespread waterlogging and salinity problems with serious adverse impacts on agricultural production. Under the Salinity Control and Reclamation Programme (SCARP), initiated in the early 1960s, vertical subsurface drains, i.e.

deep tubewells, were installed in large tracts of affected lands, while horizontal subsurface drainage has been used for smaller areas.

As a result of the low water allowances and the corresponding low design cropping intensities, there has been constant demand for more irrigation water in order to cultivate the available farmland more intensively. This demand has continued to grow with the increasing population and land pressure. Alongside the need to implement water conservation measures to increase farmgate water availability, drainage water represents a source of additional irrigation water.

Government policies encourage the maximum use of groundwater pumped for drainage for irrigation in conjunction with the canal supplies. Where groundwater is saline, drainage effluent is allowed to be disposed into the canals for reuse after dilution or it can be conveyed to the rivers through drains at times of high river flows. Alternative disposal measures have been provided only where the groundwater effluent is too saline.

Groundwater reuse in SCARP areas has had a significant impact on agricultural production. With the increased irrigation supplies, the cropping intensities in these areas rose from an average of 80 percent in the 1960s and early 1970s to an average of 116 percent in the mid-1980s. The drainage relief provided in conjunction with increased irrigation supplies through groundwater reuse has had a positive impact in terms of reducing salt-affected soils. However, the irrigation technology chosen may not have been the most suitable as the high O&M costs of the deep tubewells placed a heavy burden on the limited budget of the Irrigation Department. To ensure sustained drainage, tubewells in the fresh groundwater zones have been transferred to private sector undertakings, which use the effluent for irrigation purposes.

In zones where groundwater salinity exceeds 4.5 dS/m at a depth of 38 m, the effluent cannot be used for irrigation and needs to be safely disposed of. This might cause problems in certain areas. However, in large areas within the saline groundwater zones, shallow layers of usable groundwater lies on top of the saline groundwater. In these areas, skimming wells or horizontal subsurface drains could be used to provide the required drainage relief without disturbing the deeper saline groundwater. Research has also shown that the effluent of horizontal subsurface drains improves with time. Thus, the drainage effluent from skimming wells and horizontal subsurface drains could subsequently be used for irrigation, so also alleviating disposal problems.

The case study also focuses on the reuse of water whose quality would be regarded as poor or marginal. A number of local research institutes have investigated the effect of marginal and poor-quality tubewell waters on crop production and soils. Based on their findings and experiences elsewhere, a case is made for saline agriculture. It shows that a number of salt tolerant crops and grasses can be commercially grown using water with a salinity of up to 27 dS/m with only a moderate reduction in yield. Similarly, salt tolerant trees for fuel and forage production have been identified which can be irrigated with waters with a salinity of 19 dS/m. Salt bushes, which can tolerate high levels of salinity, have also been identified as a supplementary source of animal feed.

Evaporation ponds have been provided for the disposal of highly saline drainage effluent from irrigated areas bordering the desert towards the southeast of the country. These areas are located 500-800 km from the sea and they are characterized by interdunal depressions with highly sodic soils lying between longitudinal sand dunes 4-9 m high. In order to develop the evaporation ponds, dykes were provided across the saddles and channels cut across the dunes to form a series of connected ponds. Soon after the ponds became operational, some irrigated areas close to the ponds were severely affected by waterlogging. This could be due to significant seepage losses from the ponds generating a zone of high groundwater that obstructs or retards the natural subsurface drainage from the irrigated lands. In the space of four years, the affected area had grown to 4 200 ha.

A spinal drain has been constructed for areas located close to the sea. The outfall drain is 250 km long and has a capacity at the outfall of 113 m^3/s. It has been constructed to convey highly saline subsurface drainage effluent from 577 000 ha and the rainfall excess from in and around the area to the sea. As the vertical method of subsurface drainage predominates, the operational plans provide for the tubewell pumping to be stopped at times of heavy rain storms in order to make room in the tributary surface drains for the evacuation of the excess rainfall. Experience to date suggests that the lower fringes of the irrigated area are not at risk although the gradients of the drains at the outfall are as low as 1:14 000.

Drainage water reuse and disposal: a case study on the western side of the San Joaquin Valley, California, the United States of America

By *K.K. Tanji*

Summary

This case study focuses on the western side of the San Joaquin Valley, California, the United States of America. This area comprises the subareas of Northern, Grasslands, Westlands, Tulare and Kern. In the western side of the valley there is about 938 000 ha of irrigated land which receives imported canal water from the Sacramento Valley. Water delivery to irrigated agriculture is based on water rights and water availability. The principal crops are cotton, almonds, grapes, tomatoes, feed grains, alfalfa hay, sugar beets, oilseeds, onions, garlic, lettuce, melons and broccoli.

The western side of the San Joaquin Valley is affected by worsening waterlogging and salinity problems. In 1990, water tables less than 1.5 m from the land surface were found on about 20 650 ha in Grasslands, 2 020 ha in Westlands, 17 000 ha in Tulare, and 4 450 ha in Kern. The estimated volume of collected subsurface drainage waters in 1990 was 46.854 million m^3 in Grasslands, 4.932 million m^3 in Westlands, 39.456 million m^3 in Tulare, and 9.864 million m^3 in Kern. The extent of waterlogging and volume of subsurface drainage waters that need to be managed under a no-action scenario up to 2040 is expected to result in an increase of 138 percent in waterlogging and an increase of 143 percent in drainage water volume.

The soils on the western side of the valley are derived from marine sedimentary rocks of the mountains of the Coast Range and are thus naturally saline. The rise in the water table does not only cause waterlogging, it also brings salts into the rootzone. Shallow groundwater and rootzone drainage intercepted by tile drains is of poor quality. At the valley level, the salinity levels in these waters are elevated with a geometric mean of 13 073 ppm. The geometric means for boron and selenium concentrations are 14.9 ppm and 12.3 ppb, respectively. Currently, there is a salt imbalance in the western side of the valley based on the salt content of the water inflow and outflow. More salts need to be discharged out of the basin to achieve a salt balance to sustain irrigated agriculture.

The Sate of California promotes efficient water use through policies and legislation. Government policies exist for the reuse of reclaimed wastewater but not for the reuse of irrigation subsurface drainage water. However, there are constraints on the discharge of irrigation return flows to public water bodies. These constraints on drainage water discharges serve as an incentive for improved water management practices.

For the western side of the San Joaquin Valley, eight drainage water management practices were identified. As the principal constraint is subsurface drainage water disposal, the management options are broader than merely reuse and disposal. Drainage water management options include: source reduction, drainage water reuse, drainage water treatment, disposal in evaporation ponds, land retirement, groundwater management, river discharge, and salt utilization.

A study from the Grasslands subarea shows that source reduction is considered to be the preferred drainage water management option. This is because it is comparatively easy for many growers to implement, contributes to managing water more efficiently and reduces the volume of drainage water that needs to be disposed. A combination of source reduction and drainage water reuse is a natural follow-up to reduce irrigation return flows and to meet water disposal requirements for river discharge. Moreover, source reduction and drainage water reuse will reduce the volume of drainage water and thus help reduce the need for evaporation ponds, drainage water treatment, groundwater management and land retirement. However, drainage water reuse could degrade the soil physically and chemically if the loading rate is too large. Moreover, deep percolation may eventually degrade groundwater if the concentrated drainage water is not intercepted and removed. Implementing a real-time drainage water disposal programme could expand the possibilities for river disposal.

For drainage water treatment, the flow-through wetland system appears to be the most promising option. It is relatively inexpensive yet fairly effective at reducing aqueous selenium concentrations. However, this form of drainage water treatment has been implemented mainly on a pilot scale to date. Other technologies, such as reverse osmosis, do not at present appear to be more viable economically. Land retirement will set aside land for wildlife habitat and as resting areas for migratory birds. However, alternatives to land retirement (including active land management) may yield the same results and thus should be considered. When retiring land, if water formerly used for the irrigation of those lands is merely conveyed to nearby lands for irrigation, then little improvement will be achieved in terms of reducing drainage water quantity. Evaporation ponds would decrease the volume of drainage water which would otherwise be disposed into rivers. However, the mitigation measures and other measures necessary to meet the water disposal requirements might be expensive, especially where selenium concentrations are high. Although groundwater management can play a major role in addressing drainage problems, the management options available for groundwater are all of a long-term nature and may be expensive. Salt utilization offers good long-term potential for helping to meet the salt balance in the valley. However, the conditions of a profitable market and economic harvesting and transport arrangements are not in place.

The current drainage water management options being practised, such as offsite drainage water disposal, source reduction and drainage water reuse, are perceived to be inadequate to sustain irrigated agriculture in the western side of the valley. For example, less than 50 percent of the salts imported with irrigation water is discharged from the western side. Additional out-of-valley disposal of salts through ocean disposal or inland salt sinks is constrained by environmental concerns and costs. However, a concerted effort is underway to sustain agriculture with other drainage water management options. If waste discharge requirements for the disposal of drainage waters cannot be met after implementing many of the drainage water management options, it may be necessary to modify the waste discharge requirements, export evapoconcentrated drainage waters (brines) to the ocean or a designated salt sink, or limit the importation of water for irrigation. The sustainability of irrigated agriculture in the western side of the San Joaquin Valley is a public policy question.

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Annexes

Annex 1
Crop salt tolerance data

Introduction

In 1985, FAO published a revised version of Irrigation and Drainage Paper No. 29. This publication incorporated an extensive list of crop salt tolerance data. Since then, Maas and Grattan (1999) have published updated lists of salt tolerance data. This annex reproduces these data together with the introductory sections.

Crop yield response functions

The salt tolerance of a crop can best be described by plotting its relative yield as a continuous function of soil salinity. For most crops, this response function follows a sigmoidal relationship. However, some crops may die before the seed or fruit yields decrease to zero, thus eliminating the bottom part of the sigmoidal curve. Maas and Hoffman (1977) proposed that this response curve could be represented by two line segments: one, a tolerance plateau with a zero slope, and the other, a concentration-dependent line whose slope indicates the yield reduction per unit increase in salinity. The point at which the two lines intersect designates the threshold, i.e. the maximum soil salinity that does not reduce yield below that obtained under non-saline conditions. This two-piece linear response function provides a reasonably good fit for commercially acceptable yields plotted against the electrical conductivity of the saturated paste (EC_e). EC_e is the traditional soil salinity measurement with units of decisiemens per metre (1 dS/m = 1 mmho/cm). For soil salinities exceeding the threshold of any given crop, relative yield (Y_r) can be estimated with the following equation:

$$Y_r = 100 - b(EC_e - a) \tag{1}$$

where a = the salinity threshold expressed in dS/m; b = the slope expressed in percent per dS/m; and EC_e = the mean electrical conductivity of a saturated paste taken from the rootzone.

The two-piece linear response function is also reasonably accurate when salinity is expressed in terms of the osmotic potential of the soil solution at field capacity (OP_{fc}). When the OP_{fc} is known, yield responses can be determined as a function of the osmotic stress that the plants experience. For osmotic potentials exceeding the threshold of a crop:

$$Y_r = 100 - B(OP_{fc} - A) \tag{2}$$

where A = the salinity threshold expressed in bars; B = the slope expressed in percent per bar; and OP_{fc} = osmotic potential of the soil water extracted from the rootzone at field capacity. Equation 2, like Equation 1, is linear even though OP_{fc} is not a linear function of EC_e. However, the deviation from linearity is small, and relative yields calculated from Equation 2 are within 2 percent of those calculated from Equation 1. The salt tolerance data in the subsequent sections are expressed in terms of EC_e. Threshold (A) and slope (B) parameters in terms of OP_{fc} can be determined from the EC_e data with the following relationships:

$$A = -0.725a^{1.06} \tag{3}$$

$$B = \frac{100}{-0.725\frac{100+ab^{1.06}}{b} - A} \qquad (4)$$

These equations are based on the relationship, $OP_{fc} = -0.725\ EC_e^{1.06}$, which was obtained from Figure 6 of the USDA Handbook No. 60 (USSL, 1954) after converting osmotic pressure in atmospheres at 0°C to osmotic potential in bars at 25°C. It is further assumed that the soluble salt concentration in the soil water at field capacity is twice that of the saturated-soil extract.

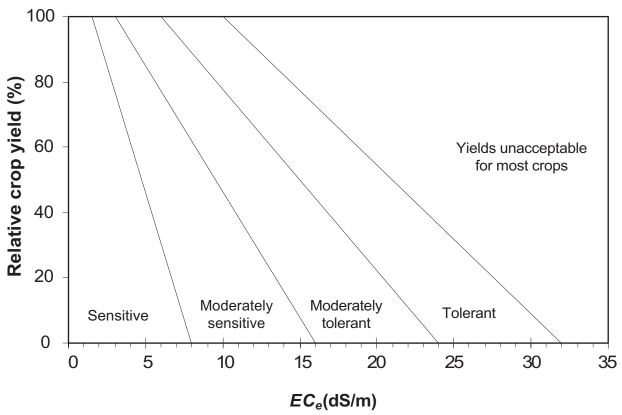
The threshold and slope concept has its greatest value in providing general salt tolerance guidelines for crop management decisions. Farmers need to know the soil salinity levels that begin to reduce yield and how much yield will be reduced at levels above the threshold. However, more precise plant response functions would be advantageous for crop simulation modelling. Van Genuchten and Hoffman (1984) have described several non-linear models that more accurately describe the sigmoidal growth response of plants to salinity. Computer programs for these models were developed and documented by Van Genuchten (1983).

SALT TOLERANCE DATA

Herbaceous crops

Table A1.1 lists threshold and slope values for 81 crops in terms of EC_e. Most of the data were obtained where crops were grown under conditions simulating recommended cultural and management practices for commercial production. Consequently, the data indicate relative tolerances of different crops grown under different conditions and not under a standardized set of conditions. Furthermore, the data apply only where crops are exposed to fairly uniform salinities from the late seedling stage to maturity. Where crops have particularly sensitive stages, the tolerance limits are given in the footnotes.

FIGURE A1.1
Division for classifying crop tolerance to salinity



The data in Table A1.1 apply to soils where chloride is the predominant anion. Because of the dissolution of $CaSO_4$ when preparing saturated-soil extracts, the EC_e of gypsiferous (non-sodic, low Mg^{2+}) soils will be 1–3 dS/m higher than that of non-gypsiferous soils having the same soil water conductivity at field capacity (Bernstein, 1962). The extent of this dissolution depends upon the exchangeable ion composition, *CEC*, and solution composition. Therefore, plants grown on gypsiferous soils will tolerate EC_es approximately 2 dS/m higher than those listed in Table A1.1. The last column provides a qualitative salt tolerance rating that is useful in categorizing crops in general terms. Figure A1.1 illustrates the limits of these categories. Some crops have only a qualitative rating because the experimental data are inadequate for calculating the threshold and slope.

Woody crops

The salt tolerance of trees, vines and other woody crops is complicated because of additional detrimental effects caused by specific ion toxicities. Many perennial woody species are susceptible to foliar injury caused by the toxic accumulation of Cl^- and/or Na^+ in the leaves. Because different cultivars and rootstocks absorb Cl^- and Na^+ at different rates, considerable variation in tolerance may occur within an individual species.

In the absence of specific-ion effects, the tolerance of woody crops, like that of herbaceous crops, can be expressed as a function of the concentration of total soluble salts or osmotic potential of the soil solution. One could expect this condition to obtain for those cultivars and rootstocks that restrict the uptake of Cl^- and Na^+. The salt tolerance data in Table A1.2 are believed to be reasonably accurate in the absence of specific-ion toxicities. Because of the cost and time required to obtain fruit yields, tolerances of several crops have been determined for vegetative growth only. In contrast to other crop groups, most woody fruit and nut crops tend to be salt sensitive, even in the absence of specific-ion effects. Only date-palm is relatively salt tolerant, whereas olive and a few others are believed to be moderately tolerant.

TABLE A1.1
Salt tolerance of herbaceous crops[†]

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	**Botanical name**[‡]		**Threshold**[§] **(EC_e)**	**Slope**	**Rating**[¶]	**References**
			dS/m	**% per dS/m**		
		Fibre, grain and special crops				
Artichoke, Jerusalem	*Helianthus tuberosus* L.	Tuber yield	0.4	9.6	MS	Newton *et al.*, 1991
Barley[#]	*Hordeum vulgare* L.	Grain yield	8.0	5.0	T	Ayars *et al.*, 1952; Hassan *et al.*, 1970a
Canola or rapeseed	*Brassica campestris* L. [syn. *B. rapa* L.]	Seed yield	9.7	14	T	Francois, 1994a
Canola or rapeseed	*B. napus* L.	Seed yield	11.0	13	T	Francois, 1994a
Chickpea	*Cicer arietinum* L.	Seed yield	--	--	MS	Manchanda & Sharma, 1989; Ram *et al.*, 1989
Corn[‡‡]	*Zea mays* L.	Ear FW	1.7	12	MS	Bernstein & Ayars, 1949b; Kaddah & Ghowail, 1964
Cotton	*Gossypium hirsutum* L.	Seed cotton yield	7.7	5.2	T	Bernstein, 1955, 1956; Bernstein & Ford, 1959a
Crambe	*Crambe abyssinica* Hochst. ex R.E. Fries	Seed yield	2.0	6.5	MS	Francois & Kleiman, 1990
Flax	*Linum usitatissimum* L.	Seed yield	1.7	12	MS	Hayward & Spurr, 1944
Guar	*Cyamopsis tetragonoloba* (L). Taub.	Seed yield	8.8	17	T	Francois *et al.*, 1990
Kenaf	*Hibiscus cannabinus* L.	Stem DW	8.1	11.6	T	Francois *et al.*, 1992
Millet, channel	*Echinochloa turnerana* (Domin) J.M. Black	Grain yield	--	--	T	Shannon *et al.*, 1981
Oats	*Avena sativa* L.	Grain yield	--	--	T	Mishra & Shitole, 1986; USSL[††]
Peanut	*Arachis hypogaea* L.	Seed yield	3.2	29	MS	Shalhevet *et al.*, 1969
Rice, paddy	*Oryza sativa* L.	Grain yield	3.0[§§]	12[§§]	S	Ehrler, 1960; Narale *et al.*, 1969; Pearson, 1959; Venkateswarlu *et al.*, 1972

TABLE A1.1 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Roselle	*Hibiscus sabdariffa* L.	Stem DW	--	--	MT	El-Saidi & Hawash, 1971
Rye	*Secale cereale* L.	Grain yield	11.4	10.8	T	Francois *et al.*, 1989
Safflower	*Carthamus tinctorius* L.	Seed yield	--	--	MT	Francois & Bernstein, 1964b
Sesame[¶¶]	*Sesamum indicum* L.	Pod DW	--	--	S	Yousif *et al.*, 1972
Sorghum	*Sorghum bicolor* (L.) Moench	Grain yield	6.8	16	MT	Francois *et al.*, 1984
Soybean	*Glycine max* (L.) Merrrill	Seed yield	5.0	20	MT	Abel & McKenzie, 1964; Bernstein *et al.*, 1955; Bernstein & Ogata, 1966
Sugar beet[##]	*Beta vulgaris* L.	Storage root	7.0	5.9	T	Bower *et al.*, 1954
Sugar cane	*Saccharum officinarum* L.	Shoot DW	1.7	5.9	MS	Bernstein *et al.*, 1966; Dev & Bajwa, 1972; Syed & El-Swaify, 1972
Sunflower	*Helianthus annuus* L.	Seed yield	4.8	5.0	MT	Cheng, 1983; Francois, 1996
Triticale	*X Triticosecale* Wittmack	Grain yield	6.1	2.5	T	Francois *et al.*, 1988
Wheat	*Triticum aestivum* L.	Grain yield	6.0	7.1	MT	Asana & Kale, 1965; Ayers *et al.*, 1952; Hayward & Uhvits, 1944
Wheat (semi-dwarf)[†††]	*T. aestivum* L.	Grain yield	8.6	3.0	T	Francois *et al.*, 1986
Wheat, Durum	*T. turgidum* L. var. *durum* Desf.	Grain yield	5.9	3.8	T	Francois *et al.*, 1986
Alfalfa	*Medicago sativa* L.	Shoot DW	2.0	7.3	MS	Bernstein & Francois, 1973; Bernstein & Ogata, 1966; Bower *et al.*, 1969; Brown & Hayward, 1956; Gauch & Magistad, 1943; Hoffman *et al.*, 1975
Alkaligrass, Nuttall	*Puccinellia airoides* (Nutt.) Wats. & Coult.	Shoot DW	--	--	T*	USSL Staff, 1954
Alkali sacaton	*Sporobolus airoides* Torr.	Shoot DW	--	--	T*	USSL Staff, 1954

TABLE A1.1 (CONTINUED)

Crop			Salt Tolerance Parameters			
Common name	Botanical name[‡]	Tolerance based on	Threshold[§] (EC_e) dS/m	Slope % per dS/m	Rating[¶]	References
Barley (forage)[#]	*Hordeum vulgare* L.	Shoot DW	6.0	7.1	MT	Dregne, 1962; Hassan *et al.*, 1970a
Bentgrass, creeping	*Agrostis stolonifera* L.	Shoot DW	--	--	MS	Youngner *et al.*, 1967
Bermudagrass[‡‡]	*Cynodon dactylon* (L.) Pers.	Shoot DW	6.9	6.4	T	Bernstein & Ford, 1959b; Bernstein & Francois, 1962; Langdale & Thomas, 1971
Bluestem, Angleton	*Dichanthium aristatum* (Poir.) C.E. Hubb. [syn. *Andropogon nodosus* (Willem.) Nash]	Shoot DW	--	--	MS*	Gausman *et al.*, 1954
Broad bean	*Vicia faba* L.	Shoot DW	1.6	9.6	MS	Ayars & Eberhard, 1960
Brome, mountain	*Bromus marginatus* Nees ex Steud.	Shoot DW	--	--	MT*	USSL Staff, 1954
Brome, smooth	*B. inermis* Leyss	Shoot DW	--	--	MT	McElgunn & Lawrence, 1973
Buffelgrass	*Pennisetum ciliare* (L). Link. [syn. *Cenchrus ciliaris*]	Shoot DW	--	--	MS*	Gausman *et al.*, 1954
Burnet	*Poterium sanguisorba* L.	Shoot DW	--	--	MS*	USSL Staff, 1954
Canarygrass, reed	*Phalaris arundinacea* L.	Shoot DW	--	--	MT	McElgunn & Lawrence, 1973
Clover, alsike	*Trifolium hybridum* L.	Shoot DW	1.5	12	MS	Ayars, 1948a
Clover, Berseem	*T. alexandrinum* L.	Shoot DW	1.5	5.7	MS	Asghar *et al.*, 1962; Ayars & Eberhard, 1958; Ravikovitch & Porath, 1967; Ravikovitch & Yoles, 1971
Clover, Hubam	*Melilotus alba* Dest. var. *annua* H.S.Coe	Shoot DW	--	--	MT*	USSL Staff, 1954
Clover, ladino	*Trifolium repens* L.	Shoot DW	1.5	12	MS	Ayars, 1948a; Gauch & Magistad, 1943

TABLE A1.1 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name‡		Threshold§ (EC_e)	Slope	Rating¶	References
			dS/m	% per dS/m		
Clover, Persian	T. resupinatum L.	Shoot DW	--	--	MS*	de Forges, 1970
Clover, red	T. pratense L.	Shoot DW	1.5	12	MS	Ayars, 1948a; Saini, 1972
Clover, strawberry	*T. fragiferum* L.	Shoot DW	1.5	12	MS	Ayars, 1948a; Bernstein & Ford, 1959b; Gauch & Magistad, 1943
Clover, sweet	*Melilotus sp*. Mill.	Shoot DW	--	--	MT*	USSL Staff, 1954
Clover, white Dutch	*Trifolium repens* L.	Shoot DW	--	--	MS*	USSL Staff, 1954
Corn (forage)††	*Zea mays* L.	Shoot DW	1.8	7.4	MS	Hassan *et al.*, 1970b; Ravikovitch, 1973; Ravikovitch & Porath, 1967
Cowpea (forage)	*Vigna unguiculata* (L.) Walp.	Shoot DW	2.5	11	MS	West & Francois, 1982
Dallisgrass	*Paspalum dilatatum* Poir.	Shoot DW	--	--	MS*	Russell, 1976
Dhaincha	*Sesbania bispinosa* (Linn.) W.F. Wight [syn. *Sesbania aculeata* (Willd.) Poir]	Shoot DW	--	--	MT	Girdhar, 1987; Karadge & Chavan, 1983
Fescue, tall	*Festuca elatior* L.	Shoot DW	3.9	5.3	MT	Bower *et al.*, 1970; Brown & Bernstein, 1953
Fescue, meadow	*Festuca pratensis* Huds.	Shoot DW	--	--	MT*	USSL Staff, 1954
Foxtail, meadow	*Alopecurus pratensis* L.	Shoot DW	1.5	9.6	MS	Brown & Bernstein, 1953
Glycine	*Neonotonia wightii* [syn. *Glycine wightii* or *javanica*]	Shoot DW	--	--	MS	Russell, 1976; Wilson, 1985
Gram, black or Urd bean	*Vigna mungo* (L.) Hepper [syn. *Phaseolus mungo* L.]	Shoot DW	--	--	S	Keating & Fisher, 1985
Grama, blue	*Bouteloua gracilis* (HBK) Lag. ex Steud.	Shoot DW	--	--	MS*	USSL Staff, 1954
Guinea grass	*Panicum maximum* Jacq.	Shoot DW	--	--	MT	Russell, 1976

TABLE A1.1 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Hardinggrass	*Phalaris tuberosa* L. var. *stenoptera* (Hack) A. S. Hitchc.	Shoot DW	4.6	7.6	MT	Brown & Bernstein, 1953
Kallargrass	*Leptochloa fusca* (L.) Kunth [syn. *Diplachne fusca* Beauv.]	Shoot DW	--	--	T	Sandhu *et al.*, 1981
Lablab bean	*Lablab purpureus* (L.) Sweet [syn. *Dolichos lablab* L.]	Shoot DW	--	--	MS	Russell, 1976
Lovegrass[§§§]	*Eragrostis sp.* N. M. Wolf	Shoot DW	2.0	8.4	MS	Bernstein & Ford, 1959b
Milkvetch, Cicer	*Astragalus cicer* L.	Shoot DW	--	--	MS*	USSL Staff, 1954
Millet, Foxtail	*Setaria italica* (L.) Beauvois	Dry matter	--	--	MS	Ravikovitch & Porath, 1967
Oatgrass, tall	*Arrhenatherum elatius* (L.) Beauvois ex J. Presl & K. Presl	Shoot DW	--	--	MS*	USSL Staff, 1954
Oats (forage)	*Avena sativa* L.	Straw DW	--	--	T	Mishra & Shitole, 1986; USSL[††]
Orchardgrass	*Dactylis glomerata* L.	Shoot DW	1.5	6.2	MS	Brown & Bernstein, 1953; Wadleigh *et al.*, 1951
Panicgrass, blue	*Panicum antidotale* Retz.	Shoot DW	--	--	MS*	Abd El-Rahman *et al.*, 1972; Gausman *et al.*, 1954
Pigeon pea	*Cajanus cajan* (L.) Huth [syn. *C. indicus* (K.) Spreng.]	Shoot DW	--	--	S	Subbaro *et al.*, 1991; Keating & Fisher, 1985
Rape (forage)	*Brassica napus* L.		--	--	MT*	USSL Staff, 1954
Rescuegrass	*Bromus unioloides* HBK	Shoot DW	--	--	MT*	USSL Staff, 1954
Rhodesgrass	*Chloris Gayana* Kunth.	Shoot DW	--	--	MT	Abd El-Rahman et al, 1972; Gausman *et al.*, 1954
Rye (forage)	*Secale cereale* L.	Shoot DW	7.6	4.9	T	Francois *et al.*, 1989
Ryegrass, Italian	*Lolium multiflorum* Lam.	Shoot DW	--	--	MT*	Shimose, 1973

TABLE A1.1 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Ryegrass, perennial	*Lolium perenne* L.	Shoot DW	5.6	7.6	MT	Brown & Bernstein, 1953
Ryegrass, Wimmera	*L. rigidum* Gaud.		--	--	MT*	Malcolm & Smith, 1971
Saltgrass, desert	*Distichlis spicta* L. var. *stricta* (Torr.) Bettle	Shoot DW	--	--	T*	USSL Staff, 1954
Sesbania	*Sesbania exaltata* (Raf.) V.L. Cory	Shoot DW	2.3	7.0	MS	Bernstein, 1956
Sirato	*Macroptilium atropurpureum* (DC.) Urb.	Shoot DW	--	--	MS	Russell, 1976
Sphaerophysa	*Sphaerophysa salsula* (Pall.) DC	Shoot DW	2.2	7.0	MS	Francois & Bernstein, 1964a
Sudangrass	*Sorghum sudanense* (Piper) Stapf	Shoot DW	2.8	4.3	MT	Bower *et al.*, 1970
Timothy	*Phleum pratense* L.	Shoot DW	--	--	MS*	Saini, 1972
Trefoil, big	*Lotus pedunculatus* Cav.	Shoot DW	2.3	19	MS	Ayars, 1948a, 1948b
Trefoil, narrowleaf birdsfoot	*L. corniculatus* var *tenuifolium* L.	Shoot DW	5.0	10	MT	Ayars, 1948a, 1948b
Trefoil, broadleaf birdsfoot	*L. corniculatus L.* var *arvenis* (Schkuhr) Ser. ex DC	Shoot DW	--	--	MS	Ayars, 1950b
Vetch, common	*Vicia angustifolia* L.	Shoot DW	3.0	11	MS	Ravikovitch & Porath, 1967
Wheat (forage)[†††]	*Triticum aestivum* L.	Shoot DW	4.5	2.6	MT	Francois *et al.*, 1986
Wheat, Durum (forage)	*T. turgidum* L. var *durum* Desf.	Shoot DW	2.1	2.5	MT	Francois *et al.*, 1986

TABLE A1.1 (CONTINUED)

Crop			Salt Tolerance Parameters			
Common name	**Botanical name**[‡]	**Tolerance based on**	**Threshold**[§] **(EC_e)**	**Slope**	**Rating**[¶]	**References**
			dS/m	**% per dS/m**		
Wheatgrass, standard crested	*Agropyron sibiricum* (Willd.) Beauvois	Shoot DW	3.5	4.0	MT	Bernstein & Ford, 1958
Wheatgrass, fairway crested	*A. cristatum* (L.) Gaertn.	Shoot DW	7.5	6.9	T	Bernstein & Ford, 1958
Wheatgrass, intermediate	*A. intermedium* (Host) Beauvois	Shoot DW	--	--	MT*	Dewey, 1960
Wheatgrass, slender	*A. trachycaulum* (Link) Malte	Shoot DW	--	--	MT	McElgunn & Lawrence, 1973
Wheatgrass, tall	*A. elongatum* (Hort) Beauvois	Shoot DW	7.5	4.2	T	Bernstein & Ford, 1958
Wheatgrass, western	*A. smithii* Rydb.	Shoot DW	--	--	MT*	USSL Staff, 1954
Wildrye, Altai	*Elymus angustus* Trin.	Shoot DW	--	--	T	McElgunn & Lawrence, 1973
Wildrye, beardless	*E. triticoides* Buckl.	Shoot DW	2.7	6.0	MT	Brown & Bernstein, 1953
Wildrye, Canadian	*E. canadensis* L.	Shoot DW	--	--	MT*	USSL Staff, 1954
Wildrye, Russian	*E. junceus* Fisch.	Shoot DW	--	--	T	McElgunn & Lawrence, 1973
Vegetables and fruit crops						
Artichoke	*Cynara scolymus* L.	Bud yield	6.1	11.5	MT	Francois, 1995
Asparagus	*Asparagus officinalis* L.	Spear yield	4.1	2.0	T	Francois, 1987
Bean, common	*Phaseolus vulgaris* L.	Seed yield	1.0	19	S	Bernstein & Ayars, 1951; Hoffman & Rawlins, 1970; Magistad *et al.*, 1943; Nieman & Bernstein, 1959; Osawa, 1965
Bean, lima	*P. lunatus* L.	Seed yield	--	--	MT*	Mahmoud *et al.*, 1988
Bean, mung	*Vigna radiata* (L.) R. Wilcz.	Seed yield	1.8	20.7	S	Minhas *et al.*, 1990
Cassava	*Manihot esculenta* Crantz	Tuber yield	--	--	MS	Anonymous, 1976; Hawker & Smith, 1982

TABLE A1.1 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Beet, red[##]	*Beta vulgaris* L.	Storage root	4.0	9.0	MT	Bernstein *et al.*, 1974; Hoffman & Rawlins, 1971; Magistad *et al.*, 1943
Broccoli	*Brassica oleracea* L. (Botrytis Group)	Shoot FW	2.8	9.2	MS	Bernstein & Ayars, 1949a; Bernstein *et al.*, 1974
Brussels sprouts	*B. oleracea L.* (Gemmifera Group)		--	--	MS*	
Cabbage	*B. oleracea* L. (Capitata Group)	Head FW	1.8	9.7	MS	Bernstein & Ayars, 1949a; Bernstein *et al.*, 1974; Osawa, 1965
Carrot	*Daucus carota* L.	Storage root	1.0	14	S	Bernstein & Ayars, 1953a; Bernstein *et al.*, 1974; Lagerwerff & Holland, 1960; Magistad *et al.*, 1943; Osawa, 1965
Cauliflower	*Brassica oleracea* L. (Botrytis Group)		--	--	MS*	
Celery	*Apium graveolens* L. var *dulce* (Mill.) Pers.	Petiole FW	1.8	6.2	MS	Francois & West, 1982
Corn, sweet	*Zea mays* L.	Ear FW	1.7	12	MS	Bernstein & Ayars, 1949b
Cowpea	*Vigna unguiculata* (L.) Walp.	Seed yield	4.9	12	MT	West & Francois, 1982
Cucumber	*Cucumis sativus* L.	Fruit yield	2.5	13	MS	Osawa, 1965; Ploegman & Bierhuizen, 1970
Eggplant	*Solanum melongena* L. var *esculentum* Nees.	Fruit yield	1.1	6.9	MS	Heuer *et al.*, 1986
Garlic	*Allium sativum* L.	Bulb yield	3.9	14.3	MS	Francois, 1994b
Gram, black or Urd bean	*Vigna mungo* (L.) Hepper [syn. *Phaseolus mungo* L.]	Shoot DW	--	--	S	Keating & Fisher, 1985

TABLE A1.1 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Kale	*Brassica oleracea* L. (Acephala Group)		--	--	MS*	Malcolm & Smith, 1971
Kohlrabi	*Brassica oleracea* L. (Gongylodes Group)		--	--	MS*	
Lettuce	*Lactuca sativa* L.	Top FW	1.3	13	MS	Ayars *et al.*, 1951; Bernstein *et al.*, 1974; Osawa, 1965
Muskmelon	*Cucumis melo* L. (Reticulatus Group)	Fruit yield	1.0	8.4	MS	Mangal *et al.*, 1988; Shannon & Francois, 1978
Okra	*Abelmoschus esculentus* (L.) Moench	Pod yield	--	--	MS	Masih *et al.*, 1978; Paliwal & Maliwal, 1972
Onion (bulb)	*Allium cepa* L.	Bulb yield	1.2	16	S	Bernstein & Ayars, 1953b; Bernstein *et al.*, 1974; Hoffman & Rawlins, 1971; Osawa, 1965
Onion (seed)		Seed yield	1.0	8.0	MS	Mangal *et al.*, 1989
Parsnip	*Pastinaca sativa* L.		--	--	S*	Malcolm & Smith, 1971
Pea	*Pisum sativum* L.	Seed FW	3.4	10.6	MS	Cerda *et al.*, 1982
Pepper	*Capsicum annuum* L.	Fruit yield	1.5	14	MS	Bernstein, 1954; Osawa, 1965; USSL[††]
Pigeon pea	*Cajanus cajan* (L.) Huth [syn. *C. indicus* (K.) Spreng.]	Shoot DW	--	--	S	Keating & Fisher, 1985; Subbarao *et al.*, 1991
Potato	*Solanum tuberosum* L.	Tuber yield	1.7	12	MS	Bernstein *et al.*, 1951
Pumpkin	*Cucurbita pepo* L. var *Pepo*		--	--	MS*	
Purslane	*Portulaca oleracea* L.	Shoot FW	6.3	9.6	MT	Kumamoto *et al.*, 1992
Radish	*Raphanus sativus* L.	Storage root	1.2	13	MS	Hoffman & Rawlins, 1971; Osawa, 1965

TABLE A1.1 (CONTINUED)

Crop		Salt Tolerance Parameters				
Common name	**Botanical name**‡	**Tolerance based on**	**Threshold**§ **(EC_e)**	**Slope**	**Rating**¶	**References**
			dS/m	**% per dS/m**		
Spinach	*Spinacia oleracea* L.	Top FW	2.0	7.6	MS	Langdale *et al.*, 1971; Osawa, 1965
Squash, scallop	*Cucurbita pepo* L. var *melopepo* (L.) Alef.	Fruit yield	3.2	16	MS	Francois, 1985
Squash, zucchini	*C. pepo* L. var *melopepo* (L.) Alef.	Fruit yield	4.9	10.5	MT	Francois, 1985; Graifenberg *et al.*, 1996
Strawberry	*Fragaria* x *Ananassa* Duch.	Fruit yield	1.0	33	S	Ehlig & Bernstein, 1958; Osawa, 1965
Sweet potato	*Ipomoea batatas* (L.) Lam.	Fleshy root	1.5	11	MS	Greig & Smith, 1962; USSL††
Tepary bean	*Phaseolus acutifolius* Gray		--	--	MS*	Goertz & Coons, 1991; Hendry, 1918; Perez & Minguez, 1985
Tomato	*Lycopersicon lycopersicum* (L.) Karst. ex Farw. [syn. *Lycopersicon esculentum* Mill.]	Fruit yield	2.5	9.9	MS	Bierhuizen & Ploeman, 1967; Hayward & Long, 1943; Lyon, 1941; Shalhevet & Yaron, 1973
Tomato, cherry	*L. lycopersicum* var. *Cerasiforme* (Dunal) Alef.	Fruit yield	1.7	9.1	MS	Caro *et al.*, 1991
Turnip	*Brassica rapa* L. (Rapifera Group)	Storage root	0.9	9.0	MS	Francois, 1984
Turnip (greens)		Top FW	3.3	4.3	MT	
Watermelon	*Citrullus lanatus* (Thunb.) Matsum. & Nakai	Fruit yield	--	--	MS*	de Forges, 1970
Winged bean	*Psophocarpus tetragonolobus* L. DC	Shoot DW	--	--	MT	Weil & Khalis, 1986

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 (Liberty Hyde Bailey Hortorium Staff, 1976) where possible.

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

¶ Ratings are defined by the boundaries in Figure A1.1. 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 percent per dS/m above a threshold of 1.9 dS/m (Hoffman *et al.*, 1983).

§§ Because paddy rice is grown under flooded conditions, values refer to the electrical conductivity 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 percent more tolerant, and common and Greenfield are about 20 percent less tolerant than the average.

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

TABLE A1.2
SALT TOLERANCE OF WOODY CROPS[†]

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Almond	*Prunus duclis* (Mill.) D.A. Webb	Shoot growth	1.5	19	S	Bernstein *et al.*, 1956; Brown *et al.*, 1953
Apple	*Malus sylvestris* Mill.		--	--	S	Ivanov, 1970
Apricot	*Prunus armeniaca* L.	Shoot growth	1.6	24	S	Bernstein *et al.*, 1956
Avocado	*Persea americana* Mill.	Shoot growth	--	--	S	Ayars, 1950a; Haas, 1950
Banana	*Musa acuminata* Colla	Fruit yield	--	--	S	Israeli *et al.*, 1986
Blackberry	*Rubus macropetalus* Doug. ex Hook	Fruit yield	1.5	22	S	Ehlig, 1964
Boysenberry	*Rubus ursinus* Cham. and Schlechtend	Fruit yield	1.5	22	S	Ehlig, 1964
Castor seed	*Ricinus communis* L.		--	--	MS[*]	USSL Staff, 1954
Cherimoya	*Annona cherimola* Mill.	Foliar injury	--	--	S	Cooper, Cowley & Shull, 1952
Cherry, sweet	*Prunus avium* L.	Foliar injury	--	--	S[*]	Beeftink, 1955
Cherry, sand	*Prunus besseyi* L., H. Baley	Foliar injury, stem growth	--	--	S[*]	Zhemchuzhnikov, 1946
Coconut	*Cocos nucifera* L.		--	--	MT[*]	Kulkarni *et al.*, 1973
Currant	*Ribes sp.* L.	Foliar injury, stem growth	--	--	S[*]	Beeftink, 1955; Zhemchuzhnikov, 1946
Date-palm	*Phoenix dactylifera* L.	Fruit yield	4.0	3.6	T	Furr & Armstrong, 1962; Furr & Ream, 1968; Furr *et al.*, 1966

TABLE A1.2 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Fig	*Ficus carica* L.	Plant DW	--	--	MT[*]	Patil & Patil, 1983a; USSL Staff, 1954
Gooseberry	*Ribes sp.* L.		--	--	S[*]	Beeftink, 1955
Grape	*Vitis vinifera* L.	Shoot growth	1.5	9.6	MS	Groot Obbink & Alexander, 1973; Nauriyal & Gupta, 1967; Taha *et al.*, 1972
Grapefruit	*Citrus* x *paradisi* Macfady.	Fruit yield	1.2	13.5	S	Bielorai *et al.*, 1978
Guava	*Psidium guajava* L.	Shoot & root growth	4.7	9.8	MT	Patil *et al.*, 1984
Guayule	*Parthenium argentatum* A. Gray	Shoot DW Rubber yield	8.7 7.8	11.6 10.8	T T	Maas *et al.*, 1988
Jambolan plum	*Syzygium cumini* L.	Shoot growth	--	--	MT	Patil & Patil, 1983b
Jojoba	*Simmondsia chinensis* (Link) C. K. Schneid	Shoot growth	--	--	T	Tal *et al.*, 1979; Yermanos *et al.*, 1967
Jujube, Indian	*Ziziphus mauritiana* Lam.	Fruit yield	--	--	MT	Hooda *et al.*, 1990
Lemon	*Citrus limon* (L.) Burm. f.	Fruit yield	1.5	12.8	S	Cerda *et al.*, 1990
Lime	*Citrus aurantiifolia* (Christm.) Swingle		--	--	S[*]	
Loquat	*Eriobotrya japonica* (Thunb). Lindl.	Foliar injury	--	--	S[*]	Cooper & Link, 1953; Malcolm & Smith, 1971

TABLE A1.2 (CONTINUED)

Crop		Tolerance based on	Salt Tolerance Parameters			
Common name	Botanical name[‡]		Threshold[§] (EC_e)	Slope	Rating[¶]	References
			dS/m	% per dS/m		
Macadamia	*Macadamia integrifolia* Maiden & Betche	Seedling growth	--	--	MS[*]	Hue & McCall, 1989
Mandarin orange; tangerine	*Citrus reticulata* Blanco	Shoot growth	--	--	S[*]	Minessy *et al.*, 1974
Mango	*Mangifera indica* L.	Foliar injury	--	--	S	Cooper *et al.*, 1952
Natal plum	*Carissa grandiflora* (E.H. Mey.) A. DC.	Shoot growth	--	--	T	Bernstein *et al.*, 1972
Olive	*Olea europaea* L.	Seedling growth, Fruit yield	--	--	MT	Bidner-Barhava & Ramati, 1967; Taha *et al.*, 1972
Orange	*Citrus sinensis* (L.) Osbeck	Fruit yield	1.3	13.1	S	Bielorai *et al.*, 1988; Bingham *et al.*, 1974; Dasberg *et al.*, 1991; Harding *et al.*, 1958
Papaya	*Carica papaya* L.	Seedling growth, foliar injury	--	--	MS	Kottenmeier *et al.*, 1983; Makhija & Jindal, 1983
Passion fruit	*Passiflora edulis* Sims.		--	--	S[*]	Malcolm & Smith, 1971
Peach	*Prunus persica* (L.) Batsch	Shoot growth, Fruit yield	1.7	21	S	Bernstein *et al.*, 1956; Brown, Wadleigh, Hayward, 1953; Hayward *et al.*, 1946
Pear	*Pyrus communis* L.		--	--	S[*]	USSL Staff, 1954
Pecan	*Carya illinoinensis* (Wangenh.) C. Koch	Nut yield, trunk growth	--	--	MS	Miyamoto *et al.*, 1986
Persimmon	*Diospyros virginiana* L.		-	--	S[*]	Malcolm & Smith, 1971

TABLE A1.2 (CONTINUED)

Crop			Salt Tolerance Parameters			
Common name	Botanical name‡	Tolerance based on	Threshold§ (EC_e)	Slope	Rating¶	References
			dS/m	% per dS/m		
Pineapple	*Ananas comosus* (L.) Merrill	Shoot DW	--	--	MT	Wambiji & El-Swaify, 1974
Pistachio	*Pistacia vera* L.	Shoot growth	--	--	MS	Sepaskhah & Maftoun, 1988; Picchioni *et al.*, 1990
Plum; Prune	*Prunus domestica* L.	Fruit yield	2.6	31	MS	Hoffman *et al.*, 1989
Pomegranate	*Punica granatum* L.	Shoot growth	--	--	MS	Patil & Patil, 1982
Popinac, white	*Leucaena leucocephala* (Lam.) de Wit [syn. *Leucaena glauca* Benth.]	Shoot DW	--	--	MS	Gorham *et al.*, 1988; Hansen & Munns, 1988
Pummelo	*Citrus maxima* (Burm.)	Foliar injury	--	--	S*	Furr & Ream, 1969
Raspberry	*Rubus idaeus* L.	Fruit yield	--	--	S	Ehlig, 1964
Rose apple	*Syzygium jambos* (L.) Alston	Foliar injury	--	--	S*	Cooper & Gorton, 1951
Sapote, white	*Casimiroa edulis* Llave	Foliar injury	--	--	S*	Cooper *et al.*, 1952
Scarlet wisteria	*Sesbania grandiflora*	Shoot DW	--	--	MT	Chavan & Karadge, 1986
Tamarugo	*Prosopis tamarugo* Phil.	Observation	--	--	T	National Academy Sciences, 1975
Walnut	*Juglans* spp.	Foliar injury	--	--	S*	Beeftink, 1955

† 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 an EC_e about 2 dS/m higher than indicated.

¶ Ratings are defined by the boundaries in Figure A1.1. Ratings with an * are estimates.

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Annex 2

Water quality guidelines for livestock and poultry production for parameters of concern in agricultural drainage water

TABLE A2.1
Guide for the use of saline water for livestock and poultry

Soluble salt content	Rating	Uses
< 1 000 mg/litre (<1.5 dS/m)	Excellent	Excellent for all classes of livestock and poultry
1 000-3 000 mg/litre (1.5-5 dS/m)	Very satisfactory	Satisfactory for all classes of livestock. May cause temporary mild diarrhoea in livestock not accustomed to them. Those waters approaching the upper limits may cause some watery droppings in poultry.
3 000-5 000 mg/litre (5-8 dS/m)	Satisfactory for livestock Unfit for poultry	Satisfactory for livestock but may be refused by animals not accustomed to it. If sulphate salts predominate, animals may show temporary diarrhoea. Poor waters for poultry, often causing watery faeces, increased mortality and decreased growth especially in turkeys.
5 000-7 000 mg/litre (8-11 dS/m)	Limited use for livestock Unfit for poultry	This water can be used for livestock except for those that are pregnant or lactating. It may have some laxative effect and may be refused by animals until they become accustomed to it. It is unsatisfactory for poultry
7 000-10 000 mg/litre (11-16 dS/m)	Very limited use	Considerable risk for pregnant and lactating cows, horses, sheep and for the young of these species. It may be used for older ruminants or horses. Unfit for poultry and probably swine.
> 10 000 mg/litre (> 16 dS/m)	Not recommended	This water is unsatisfactory for all classes of livestock and poultry.

Source: FAO, 1985b, and Guyer, 1996.

TABLE A2.2
Recommendations for levels of toxic substances in drinking-water for livestock

Constituent	Upper limit (mg/litre)	Constituent	Upper limit (mg/litre)
Arsenic (As)	0.025[1]	Nickel (Ni)	1.0
Boron (B)	5.0	Nitrate + nitrite	
Cadmium (Cd)	0.05[2]	(NO_3-N + NO_2-N)	100
Chromium (Cr)	0.05[3]	Nitrite (NO_2-N)	10
Copper (Cu)	0.5-5.0[4]	Selenium (Se)	0.05
Lead (Pb)	0.1	Uranium (U)	0.2
Mercury (Hg)	0.01[5]	Vanadium (V)	0.1
Molybdenum (Mo)	0.5	Zinc (Zn)	24[6]

[1] Upper limit recommended by FAO is 0.2 mg/litre.
[2] Upper limit recommended by CCME is 0.08 mg/litre.
[3] Upper limit recommended by FAO is 1.0 mg/litre.
[4] 0.5 mg/litre for sheep, 1.0 mg/litre for cattle and 5.0 mg/litre for swine and poultry.
[5] Upper limit recommended by CCME is 0.003 mg/litre.
[6] Upper limit recommended by CCME is 50 mg/litre.

Source: FAO, 1985b, and CCME, 1999b.

Annex 3

Drinking-water quality guidelines for parameters of concern in agricultural drainage water

Source: WHO, 1996. Guidelines for drinking-water quality, 2nd ed. Vol. 2 Health criteria and other supporting information, p. 940-949; and WHO, 1998. Addendum to Vol. 2. p. 281-283. Geneva, World Health Organization. Summary tables. http://www.who.int/water_sanitation_health/GDWQ/Summary_tables/Sumtab.htm

TABLE A3.1
Bacteriological quality of all water intended for drinking

Organisms	Guideline value
E. coli or thermotolerant coliform bacteria	Must not be detectable in any 100-ml sample

TABLE A3.2
Inorganic constituents of health significance in drinking-water

	Guideline value (mg/litre)	Remarks
Arsenic	0.01 (P)	For excess skin cancer risk of 6 × 10^{-4}
Boron	0.5 (P)	
Cadmium	0.003	
Chromium	0.05 (P)	
Copper	2 (P)	Based on acute gastrointestinal effects
Lead	0.01	It is recognized that not all water will meet the guideline value immediately; meanwhile, all other recommended measures to reduce the total exposure to lead should be implemented
Manganese	0.5 (P)	Substance at or below the health-based guideline value may affect the appearance, taste and odour of the water.
Mercury (total)	0.001	
Molybdenum	0.07	
Nickel	0.02 (P)	
Nitrate (as NO_3^-)	50 (acute)	
Nitrite (as NO_2^-)	3 (acute)	
	0.2 (P) (chronic)	
Selenium	0.01	
Uranium	0.002 (P)	

(P) Provisional guideline value.

TABLE A3.3
Pesticides

	Guideline value (µg/litre)	Remarks
Alachlor	20	For excess risk of 10^{-5}
Aldicarb	10	
Aldrin/dieldrin	0.03	
Atrazine	2	
Bentazone	300	
Carbofuran	7	
Chlordane	0.2	
Chlorotoluron	30	
Cyanazine	0.6	
DDT	2	
1,2-dibromo-3-chloropropane	1	For excess risk of 10^{-5}
1,2-dibromoethane	0.4-15 (P)	For excess risk of 10^{-5}
2,4-dichlorophenoxyacetic acid (2,4-D)	30	
1,2-dichloropropane (1,2-DCP)	40 (P)	
1,3-dichloropropane		NAD
1,3-dichloropropene	20	For excess risk of 10^{-5}
Diquat	10 (P)	
Heptachlor and heptachlor epoxide	0.03	
Hexachlorobenzene	1	For excess risk of 10^{-5}
Isoproturon	9	
Lindane	2	
MCPA	2	
Methoxychlor	20	
Metolachlor	10	
Molinate	6	
Pendimethalin	20	
Pentachlorophenol	9 (P)	For excess risk of 10^{-5}
Permethrin	20	
Propanil	20	
Pyridate	100	
Simazine	2	
Terbuthylazine (TBA)	7	
Trifluralin	20	
2,4-DB	90	
Dichlorprop	100	
Fenoprop	9	
MCPB		NAD
Mecoprop	10	
2,4,5-T	9	

(P) Provisional guideline value

TABLE A3.4
Substances and parameters in drinking-water that may give rise to complaints from consumers

	Levels likely to give rise to consumer complaints[a]	Reasons for consumer complaints
Physical parameters		
Colour	15 TCU[b]	appearance
Taste and odour	—	should be acceptable
Temperature	—	should be acceptable
Turbidity	5 NTU[c]	appearance; for effective terminal disinfection, median turbidity = 1 NTU, single sample = 5 NTU
Inorganic constituents		
	Guideline value mg/litre	**Remarks**
Aluminium	0.2	depositions, discoloration
Ammonia	1.5	odour and taste
Chloride	250	taste, corrosion
Copper	1	staining of laundry and sanitary ware (health-based provisional guideline value 2 mg/litre)
Hardness	—	high hardness: scale deposition, scum formation low hardness: possible corrosion
Hydrogen sulphide	0.05	odour and taste
Iron	0.3	staining of laundry and sanitary ware
Manganese	0.1	staining of laundry and sanitary ware (health-based guideline value 0.5 mg/litre)
Dissolved oxygen	—	indirect effects
pH	—	low pH: corrosion
		high pH: taste, soapy feel
		preferably <8.0 for effective disinfection with chlorine
Sodium	200	taste
Sulphate	250	taste, corrosion
TDS	1 000	taste
Zinc	3	appearance, taste

[a] The levels indicated are not precise numbers. Problems may occur at lower or higher values according to local circumstances. A range of taste and odour threshold concentrations is given for organic constituents.
[b] TCU, true colour unit.
[c] NTU, nephelometric turbidity unit.

Annex 4

Impact of irrigation and drainage management on water and salt balance in the absence of capillary rise

LONG-TERM SALT EQUILIBRIUM EQUATION

Long-term water balance of the rootzone and leaching fraction

The basis for understanding the impact of irrigation and drainage management on the salt balance is the water balance of the rootzone. The water balance of the rootzone can be described with the following equation:

$$I_i + P_e + G - R - ET = \Delta W_{rz} \tag{1}$$

where:

I_i = irrigation water infiltrated which is the total applied irrigation water minus the evaporation losses and surface runoff (mm);
P_e = effective precipitation (mm);
G = capillary rise (mm);
R = deep percolation (mm);
ET = evapotranspiration (mm); and
ΔW_{rz} = change in moisture content in the rootzone (mm).

On a long-term basis, it can be assumed that the change in soil moisture storage is zero. The water balance then reads:

$$I_i + P_e - R^* - ET = 0 \tag{2}$$

where:

R^* = net deep percolation, *R-G* (mm).

Therefore, the depth of water percolating below the rootzone is the amount of water infiltrated minus the water extracted by the plant roots to meet its evaporation demands. The fraction of water percolating from the rootzone is called the leaching fraction (*LF*).

$$LF = \frac{R^*}{I_i + P_e} \tag{3}$$

Salt equilibrium equation of the rootzone with complete mixing

With each irrigation, salts are added to the rootzone and evapoconcentrated by crop *ET*. A fraction of the salts is leached below the rootzone with the net deep percolation water. After a certain period, salt accumulation in the soil will approach an equilibrium or steady-state concentration based on the salinity of the applied water and the *LF* (FAO, 1985b).

The calculation of rootzone salinity makes the following assumptions:

- the irrigation water mixes completely with the soil water;
- the exchange processes and chemical reactions which take place in the soil are not taken into consideration;
- the amount of salts supplied by rainfall and fertilizers and exported by crops are negligible; and
- a zone of shallow groundwater is created with the same average salinity concentration as the percolation water.

Under these assumptions, the salinity of the soil water is equivalent to the salinity of the water percolating below the rootzone. The salinity of the water percolating below the rootzone can be estimated from the salt balance:

$$IW\, C_{IW} = R^*\, C_{R^*} \tag{4}$$

where:
IW = infiltrated water ($I_i + P_e$) (mm);
C_{IW} = salt concentration of the infiltrated water (mg/litre); and
C_{R^*} = salt concentration of the net percolation water (mg/litre).

The salt concentration of the infiltrated water can be calculated as:

$$C_{IW} = \frac{I_i}{I_i + P_e} C_I \tag{5}$$

where:
C_I = salt concentration of the irrigation water (mg/litre).

The salinity of the percolation water can also be calculated with the following formula:

$$C_{R^*} = \frac{C_{IW}}{LF} \tag{6}$$

A strong relation exists between the salt concentration *C* of a solution expressed in milligrams per litre and the electrical conductivity (*EC*) of a solution in decisiemens per metre (1 dS/m corresponds approximately to 640 mg/litre). Therefore, the salinity of the net deep percolation, which is equivalent to the salinity of the soil water, can also be expressed as:

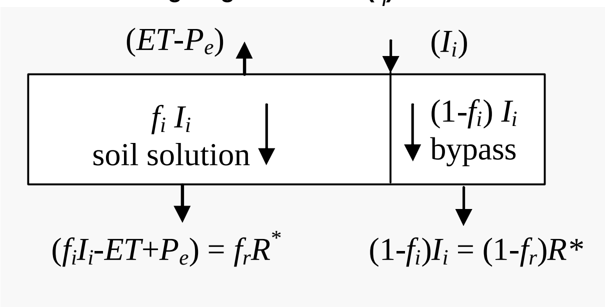
$$EC_{R^*} = EC_{SW} = \frac{EC_{IW}}{LF} \tag{7}$$

where:
EC_{SW} = electrical conductivity of the soil water (dS/m);
EC_{R^*} = electrical conductivity of the percolation water (dS/m); and
EC_{IW} = electrical conductivity of the infiltrated water (dS/m).

The EC_{SW} is inversely proportional to *LF*, i.e. a low *LF* results in a high EC_{SW} and vice versa. Under the same assumptions for *EC* as a conservative parameter, one could replace *EC* by concentrations of boron or selenium.

Salt equilibrium equation incorporating the leaching efficiency coefficient

FIGURE A4.1
Relation between leaching efficiency coefficient of the percolation water (f_r) and leaching efficiency coefficient of the incoming irrigation water (f_i)



Source: after Van Hoorn and Van Alphen, 1994.

Until now the assumption has been that all the infiltrated water mixes completely with the soil solution. In reality, a fraction of the infiltrated irrigation water percolates directly below the rootzone through cracks and macro-pores without mixing with the soil moisture solution. A more realistic estimate of the rootzone salinity can be obtained by incorporating leaching efficiency coefficients for the preferential flow pathways of salt transport. Figure A4.1 shows the relationship between the leaching efficiency coefficient of the percolation water (f_r) and the leaching efficiency coefficient related to the incoming irrigation water that mixes with the soil solution (f_i). It is assumed that P_e mixes completely with the soil solution.

The leaching efficiency coefficient of the incoming irrigation water is an independent variable determined by soil texture, structure and irrigation method, whereas the leaching efficiency coefficient of the percolation water is a dependent variable. f_r can be expressed as:

$$f_r = \frac{I_i f_i - ET + P_e}{R^*} \tag{8}$$

Integrating the leaching efficiency coefficient into the salt equilibrium equation results in:

$$EC_{f_rR^*} = EC_{SW} = \frac{EC_{IWi}}{LF_i} \tag{9}$$

in which EC_{IWi} is the salinity of the infiltrated water that mixes with the soil solution, and $EC_{frR}{}^*$ is the salinity of the percolation water which has been mixed with the soil solution. EC_{IWi} is equivalent to:

$$EC_{IWi} = \frac{f_i I_i}{f_i I_i + P_e} EC_I \tag{10}$$

The leaching fraction of the infiltrated water that mixes with the soil solution (LF_i) is:

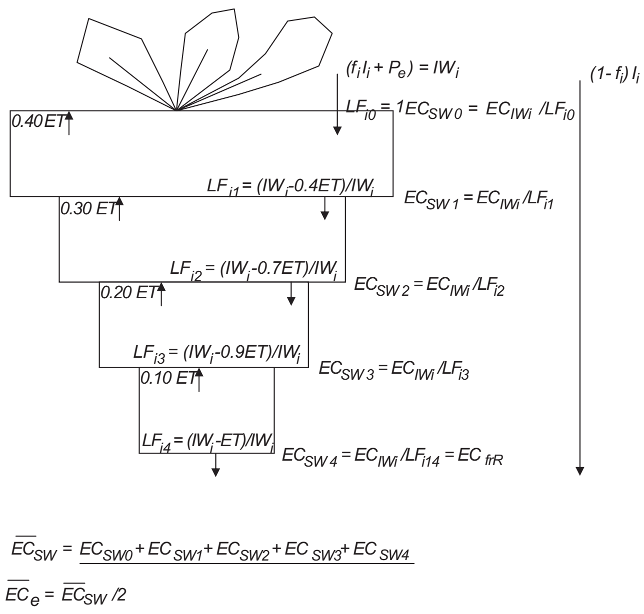
$$LF_i = \frac{f_r R^*}{f_i I_i + P_e} \tag{11}$$

Salt equilibrium equation in the rootzone considered as a four-layer profile

It is often assumed that the salinity of the net deep percolation water is equivalent to the average soil salinity (as in the previous sections). However, due to irrigation and rootwater extraction patterns, the salinity in the upper portions of the rootzone is lower than the average due to a higher *LF* (zone of salt leaching), and the salinity in the bottom portions is higher because of a smaller *LF* (zone of salt accumulation). Under normal irrigation and rooting pattern, the typical extraction pattern for the rootzone is 40-30-20-10 percent water uptake from the upper to the lower quarter of the rootzone. Where irrigation is applied more frequently, crops tend to extract more water from the upper rootzone and less from the lower rootzone. Under these conditions, the rootzone is generally shallower and the extraction pattern might be 60-30-7-3 (FAO, 1985b).

As shown in Figure A4.2, Equation 9 can also be used to calculate the rootzone salinity of five successive depths under this water uptake pattern to obtain finally the average salinity in the rootzone ($\overline{EC}_{SW}$). In most soils, the salinity of the soil water at field capacity is about twice the salinity of the soil water measured on the saturated paste (EC_e).

FIGURE A4.2
Calculation of rootzone salinity of five successive depths



It is not necessary to divide the rootzone into four equal parts (quadrants). The model can be extended into n-number of layers provided that the rootwater extraction pattern is known, e.g. the rootzone can be divided into 15-cm depth increments for 90-cm rooting depth.

Maintaining a favourable salt balance

A major concern in agricultural drainage water management is the buildup of salts and other trace elements in the rootzone to such an extent that it interferes with optimal crop growth. Applying more water than needed during the growing season for evapotranspiration can leach the salts. In areas with insufficient natural drainage, leaching water will need to be removed through artificial drainage.

Where the crop tolerance to salinity and the salinity of the irrigation water are known, the leaching requirement (LR) can be calculated. Rhoades (1974) and Rhoades and Merrill (1976) developed an empirical equation to calculate the LR:

$$LR = \frac{EC_{IW}}{5 * EC_{ts} - EC_{IW}} \tag{12}$$

EC_{ts} is the threshold salinity for a crop in decisiemens per metre of the extract from the saturated soil paste above which the yield begins to decline (Annex 1). In Equation 12, EC_{ts} represents the average rootzone salinity, and the value 5 was obtained empirically (FAO, 1985b).

If all the infiltrated water mixes completely with the soil moisture, the relation between the depth of applied water (AW) for consumptive use and the LR during a cropping season is:

$$AW = \frac{ET}{1 - LR} = I_i + P_e \tag{13}$$

However, under normal conditions a fraction of the infiltrated irrigation water, equivalent to $(1\text{-}f_i)\ I_i$, will percolate directly below the rootzone through cracks and macro-pores without mixing with the soil moisture solution. This water does not contribute to the leaching of salts from the rootzone. Under these conditions, the *LR* is:

$$LR_i = \frac{EC_{IWi}}{5 * EC_{ts} - EC_{IWi}} \tag{14}$$

The total amount of applied water is then:

$$AW = \left(\frac{ET}{1 - LR_i}\right) + (1 - f_i)I_i = (f_i I_i + P_e) + (1 - f_i)I_i \tag{15}$$

FAO (1985b) takes a different approach to assessing the *LR* for non-cracking soils. The average rootzone salinity for the four-layer concept can be calculated according to the procedures presented in Figure A4.2 in which f_i is assumed to be 1. The concentration of the salts in the rootzone varies with the *LF*. Table A4.1 shows the concentration factors for the average predicted rootzone salinity (EC_e) for a selected number of *LF*. The concentration factors can be calculated in principle for any *LF*.

TABLE A4.1
Concentration factors to predict the average *ECe* for selected leaching fractions

Leaching fraction *LF*	Concentration factor *X*
0.05	3.2
0.10	2.1
0.15	1.6
0.20	1.3
0.25	1.2
0.30	1.0
0.40	0.9
0.50	0.8
0.60	0.7
0.70	0.6
0.80	0.6

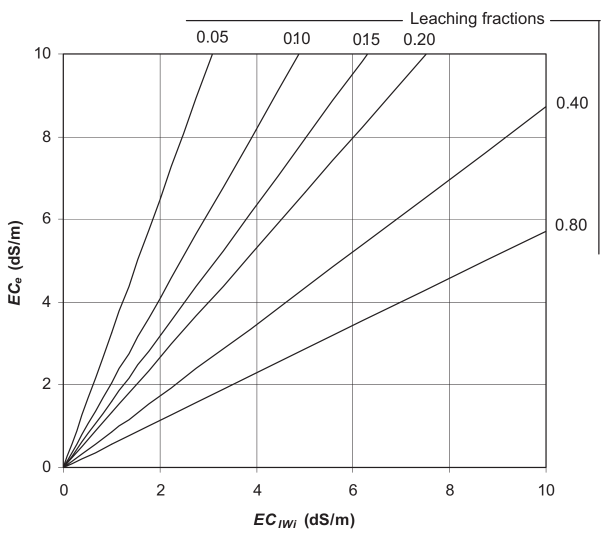
Source: FAO, 1985b.

These concentration factors can be used to calculate the relationship between EC_e and EC_{IWi} in Figure A4.3. Where the salinity of the infiltrated water and the crop tolerance to salinity are known, the necessary *LF* can be estimated from this figure. If the y-axis of the figure were EC_{ts}, then the diagonal lines would give a range of *LR*. Hence, *LF* and *LR* are commonly used interchangeably.

SALT STORAGE EQUATIONS

In previous sections, long-term steady-state conditions were assumed to prevail. To study the impact of irrigation and drainage measures on crop performance, it is important to know the changes in rootzone salinity during a cropping season over multiple time periods as well. The salt storage equation of Van Hoorn and Van Alphen (1994) can be used for such dynamic changes. If the same assumptions are made as for the steady-state equations, i.e. a zone of shallow groundwater is created with the same salinity as the percolation water, exchange processes and mineral dissolution and precipitation are not taken into consideration and

FIGURE A4.3
Assessment of leaching fraction in relation to the salinity of the infiltrated water



Source: after FAO, 1985b.

it is further assumed that the amount of salts supplied by rainfall and fertilizers and exported by crops are negligible, and the irrigation water mixes completely with the soil solution, then the salt balance for the rootzone can be described with the following equation:

$$\Delta S = S_{IW} - S_{R^*} \tag{16}$$

where:
S_{IW} = salts in infiltrated water (ECmm);
S_R^* = salts in net percolation water from the rootzone (ECmm); and
ΔS = change in salt storage in the rootzone (ECmm).

The quantity ECmm requires some explanation. The parameter *S* is the mass of salts obtained from the product of salt concentration and water volume per area. For the sake of convenience, Van Hoorn and Van Alphen (1994) chose to use *EC* instead of *TDS* in grams per litre. The unit millimetre equals litre per square metre. Thus, the parameter *S* corresponds with the amount of salt in grams per square metre. Equation 16 can also be expressed as:

$$\Delta S = IW\ EC_{IW} - R^*\ EC_{SW} \tag{17}$$

The mass of salts at the start of a period and at the end of a period normally differ and can be expressed as:

$$\Delta S = S_{end} - S_{start} \tag{18}$$

where:
S_{start} = quantity of salts in the rootzone at the start of the period (ECmm); and
S_{end} = quantity of salts in the rootzone at the end of the period (ECmm).

The salinity in the rootzone can be expressed as the conductivity of soil water or of the saturated paste. As for most soils the soil moisture content of the saturated paste is twice the soil moisture content at field capacity, the salinity at field capacity and of the saturated paste can be expressed as:

$$EC_{SW} = \frac{S}{W_{fc}} \quad \text{and} \quad EC_e = \frac{S}{2W_{fc}} \tag{19}$$

where:
S = quantity of salts in rootzone (ECmm); and
W_{fc} = moisture content at field capacity (mm).

The average salinity of the soil water during a calculation period is:

$$EC_{SW} = \frac{S_{start} + S_{end}}{2W_{fc}} = \frac{S_{start}}{W_{fc}} + \frac{\Delta S}{2W_{fc}} \tag{20}$$

Substituting Equation 20 for EC_{SW} in Equation 17 yields the salt storage equation:

$$\Delta S = \frac{IW\ EC_{IW} - \dfrac{R^* S_{start}}{W_{fc}}}{1 + \dfrac{R^*}{2W_{fc}}} \tag{21}$$

Equation 21 can be used to calculate changes in soil salinity within a cropping season. The salt storage equation can also be applied to the four-layer concept. For the calculation of the change in rootzone salinity in the first quarter, the equation becomes:

$$\Delta S_1 = \frac{IW\ EC_{IW} - \dfrac{R_1^* S_{1start}}{W_{1fc}}}{1 + \dfrac{R_1^*}{2W_{1fc}}} \tag{22}$$

In the subsequent rootzone quarters, the change in salt storage can be calculated as:

$$\Delta S_{2-4} = \frac{R_{1-3} EC_{1-3} - \dfrac{R_{2-4}^* S_{2-4start}}{W_{2-4fc}}}{1 + \dfrac{R_{2-4}^*}{2W_{2-4fc}}} \tag{23}$$

where:

1 to 4 = suffixes denoting the four quarters of the rootzone.

Integration of the leaching efficiency coefficients in the salt storage equation results in the following equation:

$$\varDelta S = \frac{(f_i I_i + P_e) EC_{IWi} - \dfrac{f_r R^* S_{start}}{W_{fc}}}{1 + \dfrac{f_r R^*}{2W_{fc}}} \tag{24}$$

Annex 5

Capillary rise and data set for soil hydraulic functions

CAPILLARY RISE

To calculate the capillary recharge, it is possible to apply Darcy's Law, which can be written for the unsaturated zone as:

$$q = -K(\theta)\left(\frac{\delta h}{\delta z}+1\right) \tag{1}$$

where:

q	= soil water flux (positive upward) (cm/d);
$K(\theta)$	= unsaturated hydraulic conductivity (cm/d);
h	= soil pressure head (cm); and
z	= vertical coordinate (positive upward) (cm).

Water balance considerations of an infinitely small soil volume result in the continuity equation for soil water:

$$\frac{\delta \theta}{\delta t} = -\frac{\delta q}{\delta z} \tag{2}$$

Combination of Equations 1 and 2 results in the general equation for flow through the unsaturated soil:

$$\frac{\delta \theta}{\delta t} = \frac{\delta\left[K(\theta)\left(\frac{\delta h}{\delta z}+1\right)\right]}{\delta z} \tag{3}$$

To calculate the capillary rise, the unsaturated hydraulic conductivity, $K(\theta)$, needs to be known. The hydraulic conductivity is a function of the moisture content and the moisture content is a function of the pressure head. The soil-water retention curve is the graph representing the relationship between pressure head and water content. Each soil has a unique soil-water retention curve. Wösten *et al*. (2001) published a series of physical soil characteristics for functional soil physical horizons (Tables A5.1, A5.2 and A5.3). These were based on measured values of soil-water retention and hydraulic conductivity. This information can be extrapolated with caution to other soil layers that have identical soil textures. In a similar manner, Mualem (1976) developed a catalogue of hydraulic properties of soils.

The unsaturated hydraulic conductivity can also be calculated based on a model developed by Van Genuchten (1980) in which Mualem's model is combined with an empirical S-shaped

Table A5.1
Dutch nomenclature known as Staring Series based on texture, organic matter content, and sand fraction - topsoils

	Dutch nomenclature[1]	Clay-silt (< 50 mm) (%)	Clay (< 2 mm) (%)	Organic matter (%)	M50 (mm)	Number curves (-)
Topsoils						
Sand						
B1	Fine to moderately fine sand	0-10		0-15	105-210	32
B2	Loamy sand	10-18		0-15	105-210	27
B3	Sandy loam	18-33		0-15	105-210	14
B4	Sandy clay loam	35-50		0-15	105-210	9
B5	Coarse sand			0-15	210-2 000	26
Silt						
B7	Silt		8-12	0-15		6
B8	Light silt loam		12-18	0-15		43
B9	Heavy silt loam		18-25	0-15		29
Clay						
B10	Silty clay loam		25-35	0-15		12
B11	Silty clay		35-50	0-15		13
B12	Clay		50-100	0-15		9
Loam						
B13	Loam	50-85		0-15		10
B14	Silt loam	85-100		0-15		67

[1]Translation of the Dutch nomenclature by means of FAO textural classes based on the clay, silt and sand fractions (FAO, 1990a).
Source: Wösten *et al.*, 2001.

Table A5.2
Dutch nomenclature known as Staring Series based on texture, organic matter content, and sand fraction - subsoils

	Dutch nomenclature[1]	Clay-Silt (< 50 mm) (%)	Clay (< 2 mm) (%)	Organic matter (%)	M50 (mm)	Number curves (-)
Subsoils						
Sand						
O1	Fine to moderately fine sand	0-10		0-3	105-210	109
O2	Loamy sand	10-18		0-3	105-210	14
O3	Sandy loam	18-33		0-3	105-210	23
O4	Sandy clay loam	33-50		0-3	105-210	9
O5	Coarse sand			0-3	210-2 000	17
Silt						
O8	Silt		8-12	0-3		14
O9	Light silt loam		12-18	0-3		30
O10	Heavy silt loam		18-25	0-3		25
Clay						
O11	Silty clay loam		25-35	0-3		11
O12	Silty clay		35-50	0-3		25
O13	Clay		50-100	0-3		19
Loam						
O14	Loam	50-85		0-3		9
O15	Silt loam	85-100		0-3		53

[1]Translation of the Dutch nomenclature by means of FAO textural classes based on the clay, silt and sand fractions (FAO, 1990a).
Source: Wösten *et al.*, 2001.

curve for the soil-water retention function to predict the unsaturated hydraulic conductivity curve. The Van Genuchten model contains six unknowns and can be written as follows:

$$K(\theta) = \frac{\left[\left(1+|\alpha h|^n\right)^m - |\alpha h|^{n-1}\right]^2}{\left[1+|\alpha h|^n\right]^{m(\lambda+2)}} K_s \tag{4}$$

where:

K_s = saturated hydraulic conductivity (cm/d);
h = pressure head (cm);
λ = dimensionless shape parameter depending on d*K*/d*h* (-);
α = shape parameter (cm^{-1});
n = dimensionless shape parameter (-); and
m = 1-1/*n* (-).

Wösten *et al*. (2001) have also published parameters for the different soil types (Table A5.4).

Although water movement in the unsaturated zone is in reality unsteady, calculations can be simplified by assuming steady-state flow during a certain period of time. The steady-state flow equation can be written as:

$$q = -K(\bar{\theta})\left(\frac{h_2 - h_1}{z_2 - z_1} + 1\right) \tag{5}$$

Equation 5 can be transformed to:

$$z_2 = z_1 + \left(\frac{h_2 - h_1}{-1 - \frac{q}{K\left(\bar{\theta}\right)}}\right) \tag{6}$$

Where $K(\bar{\theta})$ is the hydraulic conductivity for $\bar{h}$ which is $(h_1 + h_2)/2$. With this equation, the soil pressure head profiles for stationary capillary rise fluxes can be calculated. From these pressure head profiles, the contribution of the capillary rise under shallow groundwater table management can be estimated. The following section provides an example to show the calculation procedures to derive at the soil pressure head profiles for a given stationary capillary flux.

EXAMPLE TO CALCULATE THE PRESSURE HEAD PROFILES FOR A SILTY SOIL FOR STATIONARY CAPILLARY RISE FLUXES

Given is a uniform silty soil with a clay percentage of 10 percent and a low organic matter content (< 3 percent). As in semi-arid climates, the organic matter content is in general low Table A5.2 for subsoils will be used to establish the nomenclature. Table A5.2 shows that this soil can be classified as O8. Six capillary fluxes will be calculated: q = 10 mm/d; q = 7.5 mm/d; q = 5 mm/d; q = 2.5 mm/d; q = 1 mm/d; and q = 0 mm/d.

At the water table z = 0 and h = 0. To calculate the pressure head, profiles steps of h = -10 cm will be used. In the first step, h_1 = 0, h_2 = -10 cm and $\bar{h}$ = -5 cm. The parameters found in Table A5.3 and A5.4 can be used to calculate $K(\bar{\theta})$ with the model developed by Van Genuchten

TABLE A5.3
Unsaturated hydraulic conductivity $K(\theta)$ (cm/d) and θ (cm^3/cm^3) in relation to the soil pressure head $|h|$ (cm) and pF for topsoils and subsoils

$\|h\|$ (cm)	1	10	20	31	50	100	250	500	1000	2500	5000	10000	16000
pF	0.0	1.0	1.3	1.5	1.7	2.0	2.4	2.7	3.0	3.4	3.7	4.0	4.2
Topsoils													
B1 $K(\theta)$	23.41	11.38	6.04	3.13	1.14	1.6E-1	7.5E-3	6.5E-4	5.4E-4	2.0E-6	1.6E-7	1.4E-8	2.6E-9
θ	0.430	0.417	0.391	0.356	0.302	0.210	0.118	0.077	0.053	0.036	0.029	0.025	0.024
B2 $K(\theta)$	12.52	3.18	1.57	0.85	0.38	9.2E-2	1.1E-2	2.1E-3	3.8E-4	4.0E-5	7.3E-6	1.3E-6	4.2E-7
θ	0.420	0.402	0.377	0.350	0.311	0.248	0.172	0.130	0.098	0.070	0.056	0.045	0.040
B3 $K(\theta)$	15.42	6.56	4.05	2.58	1.33	3.5E-1	3.6E-2	5.2E-3	6.9E-4	4.7E-5	6.1E-6	7.9E-7	2.0E-7
θ	0.460	0.452	0.439	0.423	0.393	0.329	0.232	0.171	0.125	0.085	0.065	0.051	0.044
B4 $K(\theta)$	29.22	8.49	4.86	2.97	1.48	4.0E-1	4.4E-2	7.0E-3	1.0E-3	8.1E-5	1.2E-5	1.7E-6	4.4E-7
θ	0.460	0.451	0.438	0.423	0.397	0.345	0.263	0.208	0.163	0.119	0.095	0.077	0.067
B5 $K(\theta)$	52.91	17.54	5.97	2.14	0.54	5.6E-2	2.3E-3	2.0E-4	1.8E-5	6.9E-7	6.0E-8	5.2E-9	1.0E-9
θ	0.360	0.329	0.272	0.219	0.159	0.094	0.046	0.029	0.020	0.014	0.012	0.011	0.011
B7 $K(\theta)$	14.07	1.78	0.93	0.55	0.28	8.3E-2	1.3E-2	2.9E-3	6.0E-4	7.4E-5	1.5E-5	3.1E-6	1.0E-6
θ	0.400	0.390	0.379	0.367	0.350	0.315	0.263	0.224	0.190	0.151	0.127	0.107	0.095
B8 $K(\theta)$	2.36	0.59	0.38	0.27	0.17	7.0E-2	1.7E-2	4.9E-3	1.4E-3	2.4E-4	6.3E-5	1.7E-5	6.7E-6
θ	0.430	0.425	0.419	0.412	0.399	0.370	0.314	0.268	0.225	0.176	0.146	0.122	0.108
B9 $K(\theta)$	1.54	0.55	0.39	0.29	0.20	9.5E-2	2.6E-2	8.1E-3	2.3E-3	4.0E-4	1.0E-4	2.7E-5	1.1E-5
θ	0.430	0.427	0.423	0.418	0.409	0.385	0.331	0.280	0.229	0.173	0.138	0.111	0.095
B10 $K(\theta)$	0.70	0.14	0.10	0.07	0.05	2.4E-2	7.8E-3	2.9E-3	1.0E-3	2.4E-4	8.1E-5	2.7E-5	1.3E-5
θ	0.430	0.427	0.424	0.420	0.414	0.398	0.362	0.327	0.289	0.243	0.212	0.185	0.169
B11 $K(\theta)$	4.53	0.15	0.08	0.05	0.03	1.2E-2	3.3E-3	1.2E-3	4.0E-4	9.5E-5	3.2E-5	1.1E-5	5.2E-6
θ	0.590	0.581	0.573	0.565	0.553	0.529	0.490	0.459	0.428	0.389	0.362	0.336	0.320
B12 $K(\theta)$	5.37	0.12	0.06	0.04	0.02	7.7E-3	1.9E-3	6.3E-4	2.0E-4	4.5E-5	1.4E-5	4.6E-6	2.1E-6
θ	0.540	0.531	0.523	0.516	0.505	0.485	0.453	0.427	0.402	0.370	0.348	0.327	0.313
B13 $K(\theta)$	12.98	5.86	4.13	3.05	1.98	8.4E-1	1.8E-1	4.5E-2	1.0E-2	1.4E-3	3.0E-4	6.4E-5	2.3E-5
θ	0.420	0.417	0.411	0.403	0.390	0.354	0.280	0.220	0.168	0.117	0.089	0.068	0.057
B14 $K(\theta)$	0.80	0.29	0.21	0.16	0.11	5.0E-2	1.2E-2	2.7E-3	5.4E-4	5.4E-5	9.2E-6	1.5E-6	4.5E-7
θ	0.420	0.418	0.415	0.412	0.405	0.388	0.345	0.300	0.253	0.197	0.162	0.133	0.117
Subsoils													
O1 $K(\theta)$	15.22	11.17	6.88	3.64	1.15	9.6E-2	1.8E-3	7.6E-5	3.2E-6	5.0E-8	2.2E-9	1.2E-9	1.0E-9
θ	0.360	0.354	0.332	0.296	0.229	0.124	0.048	0.026	0.016	0.012	0.011	0.010	0.009
O2 $K(\theta)$	12.68	7.60	4.38	2.35	0.84	1.0E-1	3.2E-3	2.0E-4	1.2E-5	2.9E-7	1.7E-8	1.1E-9	1.0E-9
θ	0.380	0.372	0.351	0.321	0.269	0.179	0.092	0.058	0.040	0.028	0.024	0.022	0.021
O3 $K(\theta)$	10.87	5.71	3.48	2.10	0.96	1.9E-1	1.2E-2	1.2E-3	1.1E-4	4.8E-6	4.5E-7	4.2E-8	8.3E-9
θ	0.340	0.334	0.321	0.303	0.271	0.206	0.123	0.80	0.053	0.032	0.024	0.018	0.016
O4 $K(\theta)$	9.86	3.93	2.34	1.44	0.71	1.7E-1	1.6E-2	2.1E-3	2.7E-4	1.7E-5	2.0E-6	2.4E-7	5.8E-8
θ	0.350	0.343	0.332	0.318	0.295	0.244	0.170	0.124	0.090	0.060	0.045	0.034	0.029
O5 $K(\theta)$	25.00	10.08	2.43	0.55	0.08	3.2E-3	4.3E-5	1.6E-6	6.0E-8	8.3E-9	4.1E-9	1.2E-9	1.1E-9
θ	0.320	0.287	0.212	0.147	0.089	0.042	0.019	0.014	0.011	0.010	0.009	0.008	0.007
O8 $K(\theta)$	9.08	2.33	1.39	0.90	0.49	1.6E-1	2.6E-2	5.4E-3	1.1E-3	1.2E-4	2.3E-5	4.3E-6	1.4E-6
θ	0.470	0.462	0.451	0.438	0.417	0.372	0.269	0.239	0.191	0.140	0.111	0.088	0.075
O9 $K(\theta)$	2.23	0.86	0.58	0.41	0.26	1.0E-1	2.1E-2	5.1E-3	1.1E-3	1.5E-4	3.2E-5	6.7E-6	2.3E-6
θ	0.460	0.455	0.448	0.439	0.422	0.382	0.303	0.240	0.185	0.130	0.098	0.075	0.062
O10 $K(\theta)$	2.12	0.45	0.29	0.20	0.12	5.0E-2	1.2E-2	3.2E-3	8.4E-4	1.3E-4	3.3E-5	8.1E-6	3.1E-6
θ	0.480	0.475	0.469	0.461	0.449	0.419	0.363	0.315	0.269	0.216	0.183	0.155	0.138
O11 $K(\theta)$	13.79	0.79	0.41	0.25	0.13	4.2E-2	7.9E-3	2.0E-3	4.8E-4	7.3E-5	1.7E-5	4.0E-6	1.5E-6
θ	0.420	0.412	0.404	0.397	0.385	0.362	0.325	0.295	0.267	0.233	0.210	0.189	0.176
O12 $K(\theta)$	1.02	0.11	0.07	0.05	0.03	1.3E-2	3.8E-3	1.4E-3	4.6E-4	1.0E-4	3.4E-5	1.1E-5	5.1E-6
θ	0.560	0.555	0.550	0.544	0.534	0.512	0.470	0.431	0.392	0.342	0.308	0.278	0.259
O13 $K(\theta)$	4.37	0.10	0.05	0.03	0.02	7.6E-3	2.0E-3	6.6E-4	2.2E-4	4.8E-5	1.5E-5	5.0E-6	2.3E-6
θ	0.570	0.563	0.556	0.550	0.540	0.521	0.490	0.464	0.439	0.406	0.382	0.360	0.346
O14 $K(\theta)$	1.51	1.28	1.15	1.03	0.86	5.6E-1	1.8E-1	4.2E-2	6.3E-3	3.7E-4	4.1E-5	4.3E-6	9.4E-7
θ	0.381	0.380	0.379	0.377	0.374	0.362	0.313	0.242	0.167	0.094	0.061	0.041	0.032
O15 $K(\theta)$	3.70	1.11	0.74	0.53	0.32	1.3E-1	2.1E-2	4.0E-3	6.3E-4	4.9E-5	6.9E-6	9.5E-7	2.5E-7
θ	0.410	0.407	0.403	0.398	0.389	0.367	0.318	0.273	0.229	0.179	0.148	0.122	0.108

Source: Wösten *et al.*, 2001.

TABLE A5.4
Data set of soil hydraulic functions described with the Van Genuchten model

	q_{res} (cm^3/cm^3)	q_{sat} (cm^3/cm^3)	K_{sat} (cm/d)	a (cm^{-1})	l (-)	n (-)
Topsoils						
Sand						
B1	0.02	0.43	23.41	0.0234	0.000	1.801
B2	0.02	0.42	12.52	0.0276	-1.060	1.491
B3	0.02	0.46	15.42	0.0144	-0.215	1.534
B4	0.02	0.46	29.22	0.0156	0.000	1.406
B5	0.01	0.36	52.91	0.0452	-0.359	1.933
Silt						
B7	0.00	0.40	14.07	0.0194	-0.802	1.250
B8	0.01	0.43	2.36	0.0099	-2.244	1.288
B9	0.00	0.43	1.54	0.0065	-2.161	1.325
Clay						
B10	0.01	0.43	0.70	0.0064	-3.884	1.210
B11	0.01	0.59	4.53	0.0195	-5.901	1.109
B12	0.01	0.54	5.37	0.0239	-5.681	1.094
Loam						
B13	0.01	0.42	12.98	0.0084	-1.497	1.441
B14	0.01	0.42	0.80	0.0051	0.000	1.305
Subsoils						
Sand						
O1	0.01	0.36	15.22	0.0224	0.000	2.286
O2	0.02	0.38	12.68	0.0213	0.168	1.951
O3	0.01	0.34	10.87	0.0170	0.000	1.717
O4	0.01	0.35	9.86	0.0155	0.000	1.525
O5	0.01	0.32	25.00	0.0521	0.000	2.374
O6	0.01	0.33	33.92	0.0162	-1.330	1.311
O7	0.01	0.51	39.10	0.0123	-2.023	1.152
Silt						
O8	0.00	0.47	9.08	0.0136	-0.803	1.342
O9	0.00	0.46	2.23	0.0094	-1.382	1.400
O10	0.01	0.48	2.12	0.0097	-1.879	1.257
Clay						
O11	0.00	0.42	13.79	0.0191	-1.384	1.152
O12	0.01	0.56	1.02	0.0095	-4.295	1.158
O13	0.01	0.57	4.37	0.0194	-5.955	1.089
Loam						
O14	0.01	0.38	1.51	0.0030	-0.292	1.728
O15	0.01	0.41	3.70	0.0071	0.912	1.298

Source: Wösten *et al.*, 2001.

(Equation 4). Table A5.3 could also be used to estimate the values of *K(q)*. However, this is less accurate as interpolation between two values of *h* is required, which is difficult due to the non-linear relation.

$$K(q) = \frac{\left[\left(1+|0.0136h|^{1.342}\right)^{1-1/1.342} - |0.0136h|^{1.342-1}\right]^2}{\left[1+|0.0136h|^{1.342}\right]^{(1-1/1.342)(-0.803+2)}} 9.08$$

For h = -5 cm, *K(q)* = 3.33 (cm/d)

With Equation 6, the corresponding z_2 value for a flux of 1 cm/d (10 mm/d) can be calculated:

$$z_2 = 0 + \left(\frac{-10+0}{-1-\frac{1}{3.33}} \right) = 7.7 \text{ cm}$$

TABLE A5.5
Tabulated *z*-values (cm) for a silty soil for stationary capillary fluxes

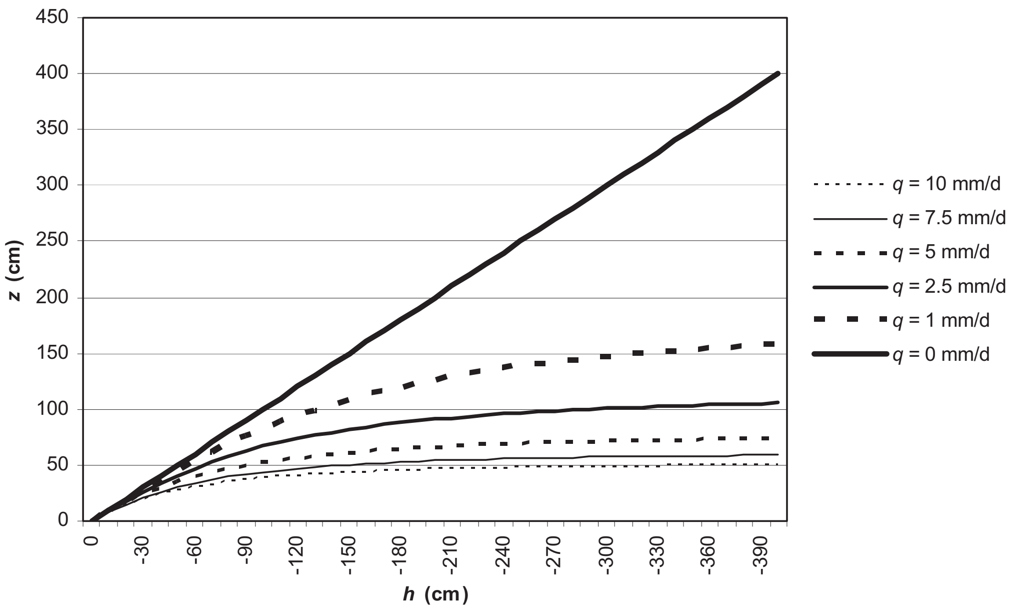
h (cm)	$\bar{h}$ (cm)	$K(\bar{?})$ (cm/d)	*z*-values (cm)					
			q = 10 mm/d	*q* = 7.5 mm/d	*q* = 5.0 mm/d	*q* = 2.5 mm/d	*q* = 1.0 mm/d	*q* = 0 mm/d
0			0	0	0	0	0	0
-10	-5	3.3	7.7	8.2	8.7	9.3	9.7	10.0
-20	-15	1.8	14.1	15.2	16.5	18.1	19.2	20.0
-30	-25	1.1	19.4	21.2	23.4	26.2	28.4	30.0
-40	-35	0.8	23.8	26.3	29.5	33.8	37.2	40.0
-50	-45	0.6	27.4	30.6	34.8	40.8	45.7	50.0
-60	-55	0.4	30.4	34.2	39.5	47.1	53.8	60.0
-70	-65	0.3	32.9	37.3	43.5	52.8	61.5	70.0
-80	-75	0.3	35.0	39.9	46.9	58.0	68.8	80.0
-90	-85	0.2	36.8	42.2	49.9	62.6	75.6	90.0
-100	-95	0.2	38.3	44.1	52.6	66.7	82.0	100.0
-110	-105	0.1	39.6	45.7	54.8	70.4	88.0	110.0
-120	-115	0.1	40.7	47.1	56.8	73.8	93.5	120.0
-130	-125	0.1	41.6	48.4	58.6	76.8	98.7	130.0
-140	-135	0.1	42.5	49.5	60.1	79.5	103.5	140.0
-150	-145	0.1	43.2	50.4	61.5	81.9	107.9	150.0
-160	-155	0.1	43.9	51.3	62.7	84.1	112.0	160.0
-170	-165	0.1	44.4	52.1	63.8	86.0	115.8	170.0
-180	-175	0.1	45.0	52.7	64.8	87.8	119.4	180.0
-190	-185	0.0	45.4	53.3	65.7	89.5	122.7	190.0
-200	-195	0.0	45.8	53.9	66.5	90.9	125.7	200.0
-210	-205	0.0	46.2	54.4	67.2	92.3	128.5	210.0
-220	-215	0.0	46.6	54.8	67.9	93.5	131.1	220.0
-230	-225	0.0	46.9	55.3	68.5	94.7	133.6	230.0
-240	-235	0.0	47.2	55.6	69.1	95.7	135.8	240.0
-250	-245	0.0	47.4	56.0	69.6	96.7	137.9	250.0
-260	-255	0.0	47.7	56.3	70.0	97.6	139.9	260.0
-270	-265	0.0	47.9	56.6	70.5	98.4	141.7	270.0
-280	-275	0.0	48.1	56.9	70.9	99.2	143.5	280.0
-290	-285	0.0	48.3	57.1	71.2	99.9	145.1	290.0
-300	-295	0.0	48.4	57.3	71.6	100.6	146.6	300.0
-310	-305	0.0	48.6	57.5	71.9	101.2	148.0	310.0
-320	-315	0.0	48.8	57.8	72.2	101.8	149.3	320.0
-330	-325	0.0	48.9	57.9	72.5	102.3	150.6	330.0
-340	-335	0.0	49.0	58.1	72.7	102.8	151.8	340.0
-350	-345	0.0	49.2	58.3	73.0	103.3	152.9	350.0
-360	-355	0.0	49.3	58.4	73.2	103.7	153.9	360.0
-370	-365	0.0	49.4	58.6	73.4	104.2	154.9	370.0
-380	-375	0.0	49.5	58.7	73.6	104.6	155.9	380.0
-390	-385	0.0	49.6	58.8	73.8	104.9	156.8	390.0
-400	-395	0.0	49.7	59.0	74.0	105.3	157.6	400.0

In the second step, h_1 = -10, h_2 = -20 cm and $\bar{h}$ = -15 cm. For $\bar{h}$ = -15 cm, $K(\bar{\theta})$ = 1.76 (cm/d). The corresponding z_2 value for a flux of 1 cm/d (10 mm/d) is:

$$z_2 = 7.7 + \left(\frac{-20+10}{-1-\dfrac{1}{1.76}} \right) = 14.1 \text{ cm}$$

In this manner, all the other values can be calculated (Table A5.5). Figure A5.1 presents the pressure head profiles as calculated.

FIGURE A5.1
Pressure head profiles for a silty soil for stationary capillary fluxes



Annex 6

Trees and shrubs for saltland, salinity ratings and species lists

The following tables and lists have been developed by the Department of Agriculture (formerly Agriculture Western Australia), Revegetation on Farms Project of the Sustainable Rural Development Program. 1998. http://www.agric.wa.gov.au/progserv/natural/trees/uses/salt2.htm

TABLE A6.1
Salinity classes for revegetation with different measures

	EC_e (dS/m)	EC_e (mS/m)	EC_{gw} (mS/m)	EM-38 hor (mS/m)	NaCl (sol. mmol/litre)	EC1:5 (w/v)[1] loam (mS/m) approx.	EC1:5 (v/v)[2] loam (mS/m) approx.
	Soil (a)	Soil (b)	Water (c)	'Soil' (d)	Water (e)	Soil	(f)
Non-saline	<2	<200	<500	<50	<20	<20	<40
Slightly	2-4	200-400	500-1 000	50-100	20-40	20-40	40-80
Moderately	4-8	400-800	1 000-2 000	100-150	40-80	40-80	80-160
Very	8-16	800-1 600	2 000-3 000	150-200	80-160	80-160	160-320
Extremely	>16	>1 600	>3 000	>200	>160	>160	>320

(a) Based on USDA 1954 categories: used by CSIRO Canberra and others in Australia.
(b) Units used in Western Australia.
(c) Groundwater from within potential rooting distance of plant (bores). Suitability for 'tree' growth.
(d) From D. Bennett and R. George, DAWA Bunbury.
(e) 'Irrigation' water used in pot trials.
(f) Based on conversions used by P. Bulman, Primary Industry SA.

[1] 1:5 (w/v) is one part by weight (g) air-dried soil to five parts by volume (ml) distilled water.
[2] 1:5 (v/v) is one volume part of soil to five volume parts of water.

TABLE A6.2
Extremely saline sites (EC_e >1 600 mS/m)

Proper Name	Common name and comments
Acacia cyclops	Coastal wattle. Severe to extreme tolerance (2, 3, 13). Sensitive to waterlogging.
Atriplex spp (A rhagodioides, A vesicaria, A paludosa)	Saltbush spp. Generally need well-drained sites.
Atriplex amnicola	River saltbush. (23, 24) Reports tolerance to 25-50 dS/m on alkaline duplex soils, and up to 38 dS/m on medium to heavy clays.
Atriplex bunburyana	Silver saltbush.
Atriplex cinerea	Grey saltbush. (23) Moderate waterlogging tolerance.
Atriplex lentiformis	Quailbrush. (24) Reports tolerance to 25-50 dS/m on alkaline duplex soils, and up to 38 dS/m on medium to heavy clays.
Atriplex muelleri	(24) Reports tolerance in subtropical and tropical areas of up to 38 dS/m on medium to heavy clays.
Atriplex nummularia	Old man saltbush. (23) Not waterlogging tolerant.
Atriplex undulata	Wavy-leaved saltbush. (24) Reports tolerance to 25-50 dS/m on alkaline duplex soils.
Acacia ampliceps	Salt wattle. (2)
Acacia stenophylla	River cooba, River myall. (2)
Casuarina glauca	Grey Buloke. (2) in 8-16 dS/m. (8, 19) Wet or dry sites. (20) Gives 50% mortality at EC 1:5 of >400 mS/m. (22)
Casuarina obesa	Salt sheoak. (6) In the 100-150 mS/m range from the EM38. (8, 9, 20)

Eucalyptus halophila	Salt lake mallee. (4, 17).
Eucalyptus kondininensis	Kondinin blackbutt. (2) (10, 11) Suggests much lower tolerance. (16, 17).
Frankenia spp. (F ambita, F brachyphylla, F fecunda)	(17).
Halosarcia spp.	Samphire. (1, 17, 20, 23) Combined waterlogging and salt tolerance is particularly high.
Melaleuca halmaturorum subsp. Cymbifolia	(4) A WA subspecies.
Melaleuca halmaturorum subsp. Halmaturorum	South Australian swamp paperbark. (2, 14, 15, 19) (24) Gives range of 15-25 dS/m.
Melaleuca thyoides	(4, 12).
Melaleuca cuticularis	Swamp paperbark, salt paperbark. (2) Suggests 8-16 dS/m. (12)
Paspalum vaginatum	Saltwater couch. (21) Very high waterlogging tolerance, no drought tolerance. Needs summer moisture.
Puccinellia ciliata	Puccinellia. (21, 23) Moderate waterlogging tolerance. (24) Reports tolerance to 25-50 dS/m on alkaline duplex soils.
Sarcocornia spp. (S quinqueflora)	Glasswort, samphire. (20) Combined salt and waterlogging tolerance is particularly high.
Sporobolus virginicus	Marine couch. (20). (24) Reports tolerance to 25-50 dS/m on alkaline duplex soils and wet sites.

TABLE A6.3
Very saline sites (EC_e 800-1 600 mS/m)

Proper name	Common name and comments
Acacia aff lineolata	(13) Good waterlogging tolerance.
Acacia brumalis	(3, 13) Sensitive to waterlogging.
Acacia cyclops	Coastal wattle. (2,3,13) See Table 2 above.
Acacia ligulata	Umbrella bush. (14)
Acacia mutabilis ssp. Stipulifera	(13)
Acacia retinodes	Wirilda. (2)
Acacia salicina	Coobah, willow wattle. (2, 14) suckers and could be invasive.
Acacia saligna	Golden wreath wattle. (2) Puts this into 4-8 dS/m. Variation in provenances. (3) (6) In the 100-150 mS/m range for EM38. (12) Very good tolerance for salt and some waterlogging.
Acacia stenophylla	River Coobah. (2, 14, 19) (24) Gives range of 15-25 dS/m.
Casuarina cristata ssp. Cristata	Black oak, Belah. (2, 8, 19)
Casuarina cristata ssp. Pauper	Belah (WA ssp.). (2)
Casuarina equisetifolia	Horsetail sheoak.
Casuarina equisetifolia var. incana	(8). Similar tolerance to Cas obesa and Cas glauca.
Eucalyptus campaspe	Silver gimlet. (2, 16)
Eucalyptus moluccana	Grey box. (2) Suggests 4-8 dS/m. (22)
Eucalyptus occidentalis	Flat top yate. (2) (6) In the 100-150 mS/m EM38 range. (10, 16, 17, 19) Wet or dry sites. (24) Range of 15-25 dS/m.
Eucalyptus raveretiana	(22) May be higher tolerance.
Eucalyptus sargentii ssp. Sargentii	Salt river gum, Sargent's mallee. (2, 10, 16, 17) Waterlogging tolerant.
Eucalyptus spathulata ssp. Spathulata	Swamp mallet. (2, 9, 10, 16)
Melaleuca decussata	Cross-leaf honey myrtle. (2, 14, 15, 19)
Melaleuca hamulosa	(12)
Melaleuca lanceolata	Rottnest Island tea tree, moonah. (2, 14, 15, 19). Needs well-drained site.
Melaleuca leucadendra	Cadjeput. (2)
Melaleuca quinquinerva	Five-veined paperbark. (2)
Melaleuca squarrosa	Scented paperbark. (2)
Melaleuca uncinata	Broombush. (2) Highly variable taxon. Variable tolerance.

TABLE A6.4
Moderately saline (EC_e 400-800 mS/m)

Proper name	Common name and comments
Acacia acuminata	Jam. (2)
Acacia collectoides	Spine wattle. (12)
Acacia iteaphylla	Flinder's Range wattle. (2)
Acacia longifolia	Sydney golden wattle. (2)
Acacia merrallii	Merrall's wattle. (12)
Acacia pendula	Weeping myall. (19)
Acacia prainii	Prain's wattle. (12)
Acacia redolens	Ravensthorpe source. (3, 12, 13) Tolerance varies with seed source.
Allocasuarina leuhmannii	Buloke. (2)
Allocasuarina verticillata	Drooping sheoak. (2, 19)
Atriplex semibaccata	Creeping saltbush.
Callistemon paludosis	River bottlebrush. (19)
Callistemon phoeniceus	Lesser bottlebrush. (12)
Casuarina cunninghamiana	River sheoak. (2, 6, 19, 22)
Chloris gayana	Rhodes grass. (21)
Eucalyptus aggregata	(2)
Eucalyptus anceps	(5)
Eucalyptus angustissima ssp. Angustissima	(5)
Eucalyptus astringens	Brown mallet. (2, 16) Seed source critical.
Eucalyptus botryoides	Southern mahogany. (2)
Eucalyptus brachycorys	Comet Vale mallee. (9)
Eucalyptus brockwayi	Dundas mahogany. (2,16)
Eucalyptus camaldulensis	River red gum. (2, 6, 9) (10) Suggests lower tolerance. (16, 17) (19, 22) Suggests higher tolerance. Provenance critical.
Eucalyptus coolabah	(2) This group is being revised. Includes E microtheca.
Eucalyptus diptera	Two-winged gimlet. (10) Suggest higher tolerance.
Eucalyptus famelica	(5)
Eucalyptus foliosa	(5)
Eucalyptus largiflorens	Black box. (2, 14, 19) Wet or dry sites.
Eucalyptus leptocalyx	Hopetoun mallee.
Eucalyptus lesouefii	Goldfields blackbutt. (16, 17)
Eucalyptus leucoxylon	South Australian blue gum. (2) Four named ssp. and highly variable. Provenance critical.
Eucalyptus leucoxylon ssp. Petiolaris	Eyre Peninsula blue gum. (17)
Eucalyptus loxophleba ssp. Lissophloia	Smooth barked York gum.
Eucalyptus loxophleba ssp. Loxophleba	York gum. (10, 16)
Eucalyptus melliodora	Yellow box. (2, 6, 22)
Eucalyptus mimica	(5) Mallet from Newdegate area.
Eucalyptus platycorys	Boorabbin mallee. (5, 9) Sensitive to waterlogging.
Eucalyptus platypus var. heterophylla	Coastal moort. (2, 10) Could have much higher tolerance.
Eucalyptus platypus var. plpatypus	Round-leafed moort.
Eucalyptus polybractea	Blue mallee. (2)
Eucalyptus rigens	(5)
Eucalyptus robusta	Swamp mahogany. (2, 6, 10, 19, 22)
Eucalyptus rudis	Flooded gum. (2, 6, 10)
Eucalyptus salicola	Salt gum.
Eucalyptus sideroxylon	Red ironbark. (19) Needs well-drained site.
Eucalyptus tereticornis	Forest red gum. (2) (22) Suggests higher tolerance.
Eucalyptus varia ssp. Salsuginosa	(5) Mallee form of E gardneri.
Eucalyptus vegrandis	(5) Syn E spathulata ssp. grandiflora.
Eucalyptus xanthonema	(5)
Festuca arundinacea	Tall fescue. (21) Moderate waterlogging tolerance.
Lagunaria patersonii	Norfolk Island hibiscus. (14, 15) Coastal.
Maireana brevifolia	Small-leaved bluebush. (23)
Melaleuca acuminata	Broombush. (12)
Melaleuca armillaris	Bracelet honey myrtle. (2, 19) Needs well-drained site.
Melaleuca bracteata	River teatree. (2)
Melaleuca brevifolia	Mallee honey myrtle. (12)
Melaleuca ericifolia	Swamp paperbark. (2, 19)

Melaleuca lateriflora	(5) Grows with M uncinata and others.
Melaleuca linarifolia	Narrow-leaved paperbark. (2)
Melaleuca microphylla	(14)
Melaleuca styphelioides	Prickly-leaved paperbark. (2, 19)
Myoporum desertii	Turkey bush. (14)
Myoporum insulare	Boobialla. (15)
Pittosporum phylliraeoides	Native apricot. (15)
Thinopyrum elongatum	Tall wheat grass. (21, 23) Moderate waterlogging tolerance
Trifolium michelianum	Balansa clover. (23) Syn. T balansae. Highly tolerant of waterlogging.

TABLE A6.5
Slightly saline (EC_e 200-400 mS/m)

Proper name	Common name and comments.
Acacia mearnsii	Late black wattle. (2) Possible weed spp.
Acacia melanoxylon	Tasmanian blackwood. (2)
Callistemon salignus	Willow bottlebrush
Casuarina littoralis	(8)
Casuarina stricta	(8)
Casuarina torulosa	(8)
Cynodon dactylon	Couch. (21)
Eucalyptus aggregata	Black gum. (21)
Eucalyptus calycogona ssp. calycogona	(5, 16)
Eucalyptus camphora	Swamp gum. (2)
Eucalyptus celastroides ssp. celastroides	Mealy blackbutt. (5)
Eucalyptus cinerea	Argyle apple. (2)
Eucalyptus cladocalyx	Sugar gum. (2)
Eucalyptus clelandii	Cleland's blackbutt. (11) Suggests higher tolerance.
Eucalyptus concinna	Victoria Desert mallee. (16, 17)
Eucalyptus conferruminata	Bald Island marlock. (17)
Eucalyptus cornuta	Yate. (2) (17) Suggests no tolerance of salt.
Eucalyptus crenulata	Victorian silver gum. (2)
Eucalyptus elata	River peppermint. (2)
Eucalyptus flocktoniae	Merrit. (16) (17) Sensitive to waterlogging.
Eucalyptus forrestiana ssp. forrestiana	Fuschia mallee. (16)
Eucalyptus globulus ssp. Globulus	Blue gum. (2)
Eucalyptus grandis	Rose gum. (2) (22) suggests moderate tolerance.
Eucalyptus griffithsii	Griffith's grey gum. (16, 17)
Eucalyptus hypochlamydea ssp. ecdysiastes	(5)
Eucalyptus longicornis	Red morrell. (16)
Eucalyptus macrandra	Long-flowered marlock. (15, 16, 17)
Eucalyptus megacornuta	Warted yate.
Eucalyptus merrickiae	Goblet mallee. (16, 17)
Eucalyptus microcarpa	(6)
Eucalyptus ovata	Swamp gum. (2)
Eucalyptus ovularis	Small-fruited mallee. (16, 17)
Eucalyptus salmonophloia	Salmon gum. (10) Suggests moderate tolerance. (16)
Eucalyptus torquata	Coral gum. (16, 17)
Eucalyptus wandoo	Wandoo. (10) Suggests moderate tolerance. Seed source important.
Eucalyptus yilgarnensis	
Phalaris aquatica	Phalaris. (21)
Schinus molle var. areira	Peppeer tree. (14)
Trifolium fragiferum	Strawberry clover. (21) High waterlogging tolerance. Best on summer moisture.

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