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THE UNIVERSITY OF ALBERTA

STUDIES OF SURFACE RETURN FLOWS AND IRRIGATED SOILS IN THE BOW RIVER IRRIGATION DISTRICT

by

© RONALD WALKER McMULLIN

A THESIS
SUBMITTED TO THE FACULTY OF GRADUATE STUDIES AND RESEARCH
IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE DEGREE
OF MASTER OF SCIENCE

DEPARTMENT OF SOIL SCIENCE

EDMONTON, ALBERTA

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THE UNIVERSITY OF ALBERTA

FACULTY OF GRADUATE STUDIES AND RESEARCH

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research, for acceptance, a thesis entitled "Studies of Surface Return Flows and Irrigated Soils in the Bow River Irrigation District" submitted by Ronald Walker McMullin in partial fulfilment of the requirements for the degree of Master of Science.

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Supervisor

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Date . . [illegible] . . . .

ABSTRACT

In recent years agriculture has been cited as a major source of pollutants found in water supplies. Irrigation return flow, which stems from agricultural lands, has escaped neither the sight nor the voice of concerned scientists.

To assess the situation in southern Alberta, the quality and quantity of irrigation return flow discharged from two tracts of land in the Bow River Irrigation District was monitored over two irrigation seasons. Nutrients, total dissolved salts, several specific dissolved salts, pH, temperature, and suspended load of waters in open irrigation and drainage channels were studied at key points throughout a 2,560 ha, flood-irrigated area near Hays, Alberta, and a 1,090 ha, sprinkler-irrigated area near Vauxhall, Alberta. The Oldman and Bow Rivers were the respective receiving waters. Soils in the area were also studied, as both receivers and donors of pollutants.

Relative to the supply waters entering the areas, the discharged return flows had slightly higher phosphorus and nitrate-nitrogen concentrations. These levels were considered to be only slightly eutrophic. Sedimentation of phosphorus-bearing suspended materials resulted in a decrease to one-half the original total phosphorus concentration in the runoff waters from one site. A decrease to one-tenth the original nitrate-nitrogen concentration in surface runoff water was also noted in the 2.5 km stretch of open drain.

Dissolved salt content in the return flow can be attributed to the following: inherent quality of the supply water, salts in

subsurface drainage, and salts added by surface runoff. These are listed in decreasing order of importance.

Quality of the return flow with respect to SAR, chloride, sulphate, bicarbonate and carbonate was found to be good. Water temperatures did not reach objectionable levels either.

Compared to receiving waters, pH values of the return flow were excessive. Compared to supply water quality, pH of the return flow was improved.

Theory would suggest that suspended load concentration should be higher in the return flow from the flood-irrigated area than from the sprinkler-irrigated area, but the opposite was found for these two areas.

Soils in the study areas receive considerable additions of potential pollutants. Only a minute portion of these pollutants are carried to the neighboring rivers.

Salt imports in the supply water exceed exports in the return flow. Though this would suggest an increase in salt content of the soils, a large decrease in soluble salt content of the fall samples (1976) relative to the spring samples (1975) was measured.

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INTRODUCTION

Society's demands on water resources have never been greater and forthcoming increases in requirements are anticipated.

As more of our water resources are being utilized, more individuals are becoming involved in water quality and quantity concerns. The municipal officer is concerned about both the allotment of tax dollars that must be spent on water treatment prior to water distribution, and whether supplies are sufficient to meet the needs of an expanding population. The industrialist is concerned about the expense of utilizing poor quality water, either because of required pretreatment or low efficiency in use; and he has anxieties about the water shortages looming in the future. The agriculturist is dismayed by low yields on previously productive land, which has been permanently damaged by the salts imported in poor quality irrigation water; and he wonders whether there is enough good water to increase production. The ecologist is concerned about the shift from natural aquatic ecosystems to unstable or undesirable conditions imposed by man's activities. Those persons seeking recreational activities on, by, or in water do not enjoy physical irritations, objectionable odors and tastes, views of green "slimes", or the catches of water weeds on fishhooks that may await them. Those who look on water as a symbol of serenity -- a source of life -- are watching the gradual disintegration of their philosophies.

The quality of a water system may be deteriorated to the extent that beneficial use is impaired, that is, the water is polluted. Eight categories of water pollutants have been broadly defined by the Committee on Pollution, National Academy of Sciences, National Research

Council (reported in Mackenthun and Ingram, 1967, p. 2): "(1) domestic sewage and other oxygen-demanding wastes; (2) infectious agents; (3) plant nutrients; (4) organic chemicals such as insecticides, pesticides, and detergents that are highly toxic at low concentrations; (5) other minerals and chemicals including chemical residues, petrochemicals, salts, acids, silts and sludges; (6) sediments from land erosion; (7) radioactive substances; (8) heat." Of these, irrigated areas have the potential to contribute significant amounts of certain pollutants mentioned within categories (3), (4), (5), (6), and (8). Irrigation agriculture in the United States has been singled out by the Program Coordinator, Water Quality and Environmental Programs, Washington State Water Pollution Control Commission as one of the major contributors of pollutants to that nation's waters (Browning and Heinemann, 1970).

What is the situation in Alberta? Is irrigation agriculture degrading our water supplies?

Because of the vast potential for pollution from irrigated lands, a study was established to assess the status of pollutant discharge from an irrigation district in southern Alberta. Dr. M. Oosterveld, an Agriculture Canada research scientist at the Lethbridge Research Station is monitoring a network of eight water discharge sites from the Bow River Irrigation District to assess the state of water quality deterioration. The study herein is a sub-project of Dr. Oosterveld's, designed to delineate sources and quantities of discharged nutrients, salts, suspended materials and water. The soils in the research area were also studied -- as both recipients of pollutants and as donors of pollutants. Two of the eight major return flow watersheds were selected for detailed study; one of which was a sprinkler-irrigated area north of Vauxhall, Alberta, and the other a flood-irrigated area southeast of Hays, Alberta.

LITERATURE REVIEW

The pollution problems associated with irrigation could all be alleviated if irrigation of lands was halted. However, when consumptive use far exceeds precipitation, as is the case in the Vauxhall and Hays areas (Sonmor, 1963; Committee of the Canada Department of Agriculture, 1960), additional water is required to obtain economically viable yields.

Irrigation Return Flow

Inherent in the present methods of irrigation in the study areas are losses of diverted water: losses to a drainage network via (1) seepage -- water passing through the structural materials of reservoirs, canals, laterals, sub-laterals, and farm ditches into the surrounding geologic material; "(2) bypass water -- water which is returned to a source of supply without being applied to irrigated land; (3) deep percolation -- applied irrigation water which finds its way to the drainage system or contributes to the ground water recharge; and (4) tail water runoff -- that portion of the applied water that runs off the land surface" (Utah State University Foundation, 1969). Any of the "lost" water which re-enters a resource pool -- whether the pool is ground water or surface water -- is subject to reuse. This potentially reusable water which has found its way back into the supply system from the irrigation system is termed irrigation return flow.

The sources of return flow may be divided into two useful categories on a management basis: point sources and diffuse sources.

"Point sources enter at discrete and identifiable locations and are therefore amenable to direct quantification and measurement of their impact on the receiving water. . . Diffuse sources may be defined as those which at present can be only partially estimated on a quantitative basis and which are probably amenable only to attenuation rather than to elimination" (Ryden et al., 1973). Regulatory action to control return flow constituents must be preceded by a categorical inventory of the sources.

Pollution Problems

Return flow is valuable inasmuch as it is a source of water for the downstream user. However, the pollution problems of irrigation return flow are serious and may be damaging, even destructive.

Eutrophication

A. Ecology of eutrophication

Eutrophication, one of the problems associated with return flow, may result in pollution of downstream bodies of water. Defined simply, eutrophication is an augmentation of the nutrient levels within a water body. Waters having high nutrient levels are said to be eutrophic (Glooschenko and Dobson, 1975). This augmentation of nutrient levels may be a result of either natural or natural-plus-artificial conditions. [illegible] activities may induce and maintain artificial nutrient augment[illegible] [illegible] this is the crux of concern.

[illegible]-an[illegible] [illegible]self eutrophication is not pollution, but it may result in poll[illegible] [illegible] natural conditions a lake passes slowly through a progression of trophic (nutrient) levels: oligotrophic to mesotrophic, to eutrophic (Wilcox, 1973). This progression, or aging of the lake, occurs at a rate significant only in terms of geologic time (Mackenthun and Ingram, 1967). In the beginning stage (oligotrophic), the lake supports very little life; the limited supply of nutrients can support only a limited biomass. As nutrient levels increase, the lake becomes mesotrophic which constitutes a more productive period of the lake. In its mesotrophic stage there is an extensive, balanced aquatic biota. The final stage, or eutrophic condition, results when nutrients entering and residing within a lake are in excess of levels required to maintain the mesotrophic balance.

A eutrophic lake is not dead, but rather is very much alive! These lakes contain vast amounts of algae and aquatic plants, which may cause a wide array of problems: increases in turbidity, nuisance growths of small suspended plants or algal scums, increased area of weed beds, offensive tastes and odors associated with decaying organisms, algal windrows on beaches, filter clogging, and death of aquatic aerobes due to oxygen depletion during periods of high community respiration (Mackenthun and Ingram, 1967 ; Wilcox, 1973).

Concern is expressed not only that a lake is eutrophic, but also that society may have inadvertently imposed that condition; Lake Erie has aged an equivalent of 15,000 years within the last 50 (Wilcox, 1973)!

B. Limiting nutrients

(a) An overview

Important to the understanding of the process of eutrophication is an understanding of the "Law of the Minimum" (or "Principle of Limiting Factors"). There exist relatively fixed ratios in nutrient requirements for metabolism; for example, carbon, nitrogen, and phosphorus are utilized by algae in a ratio of approximately 100:15:1, respectively (Carpenter et al., 1969; Garmin, 1973; Verduin, 1970). In most waters there is great disproportion between the nutrient supply and the requirements for biotic growth. For example, Ca^{++} and HCO_3^- may be found in concentrations far in excess of demands, while other ions like $H_2PO_4^-$ may be present in proportionately minute amounts (Ruttner, 1952). The nutrient which is found in least supply with respect to the others in the ratio becomes the limiting factor in nutrient uptake and hence the limiting factor to growth. Any reduction in the supply of the limiting nutrient, therefore, will also reduce the amount of primary production.

C, N, P, Fe, Mo, Mg, and Na have been shown to be limiting nutrients in various waters (Hutchinson, 1969). N and P are the only elements "that have been definitely established as consistent growth-limiting factors in natural waters" (Biggar and Corey, 1969), but Keuntzel (1969) has presented conclusive evidence that C (as CO_2) is also a limiting nutrient of great importance.

(b) Phosphorus

Phosphorus is a macro-nutrient, but because of low solubility of phosphatic compounds (see Table 1), it is found in extremely low concentrations in natural waters. The Federal Water Pollution Control Committee (1968) cites values of 10 to 30 ug/l as "normal" for natural water bodies. This perhaps is the basis for Syers et al. (1973)

Table 1. Solubilities of some inorganic calcium phosphate compounds found in soil.

Compound*	Formula*	Solubility** g per 100 ml	
Monocalcium phosphate	$Ca(H_2PO_4)_2$	1.8	↑ Solubility Increases
Dicalcium phosphate	$CaHPO_4(2CaO\cdot H_2O\cdot P_2O_5)$	0.0316	
Tricalcium phosphate	$Ca_3(PO_4)_2$		
Oxy-apatite	$3Ca_3(PO_4)_2\cdot CaO$		
Hydroxy-apatite	$3Ca_3(PO_4)_2\cdot Ca(OH)_2$		
Carbonate-apatite	$3Ca_3(PO_4)_2\cdot CaCO_3$		
Fluor-apatite	$3Ca_3(PO_4)_2\cdot CaF_2$		

* Buckman and Brady (1969).

** Weast (1974).

suggesting that phosphorus is the key to retarding or even reversing eutrophication. Maintaining phosphorus concentrations at natural levels could prevent a transition from a mesotrophic environment to a eutrophic one, or even cause a transition from eutrophic to mesotrophic conditions.

Any major user of phosphorus is suspected of being a major source of that nutrient. According to Vallentyne (1974), 70 percent of the phosphorus used in the United States in 1967 was in fertilizer, 13 percent in detergents, 8 percent in animal feeds, and 9 percent in other products. Where would the search begin to unveil phosphorus sources -- agriculture! The Task Group 2610-P Report (1967) estimates that 42 percent of the phosphorus in United States water supplies come from agricultural land.

To understand the role of agriculture in supplying phosphorus to water bodies, a knowledge of P forms, dynamics of P in the soil, and mechanisms of loss from agricultural lands is necessary. In calcareous soils having pH values of 7.0 or more, typical of those in the study area, phosphates are readily fixed by reaction with calcium to form rather insoluble compounds such as apatites (Alexander and Robertson, 1968; Hsu and Jackson, 1960; Larsen, 1967). By a process termed aging these phosphate compounds become relatively insoluble; crystal growth reduces surface area per unit volume and crystal transformations tend to produce less soluble compounds. (See Table 1 for solubilities.) These low solubility values support the observation that of the normal phosphorus content of the soil, less than 1 percent is readily available for plant use (Buckman and Brady, 1969), and concentrations in soil solution seldom exceed 0.2 mg/l (Biggar and Corey, 1969).

Does this mean there is practically no phosphorus in soil? No!

In fact, soil has a vast capacity to hold (fix) phosphorus. Toth and Bear (1947) found that the average "adsorbing capacity" of 17 New Jersey soils was 41.9 t (46.2 short tons) of 20 percent superphosphate per acre. Present rates of P application in the study areas, as will be seen later, are far less than the general fixing capacities of soil for that element. Therefore, the soil will retain most of the applied P until it is removed by plants. Larsen (1967) estimates that in arid regions, soils are 0.051 percent P by weight (510 ppm) -- far from "no" phosphorus.

From the previous discussion one can conclude that even though approximately 70 percent of the P used is fertilizer phosphorus, most of this is "tied-up" as insoluble compounds in the soil; very little of the applied phosphorus (in terms of agronomic importance) will be leached by drainage waters (Carter _et al._, 1971).

Because of the insoluble nature of phosphatic compounds and the vast capacity of soil to hold phosphorus, does this mean that no phosphorus is lost from agricultural lands? Table 2 gives values of P losses at various places and under various conditions. The following conclusions are apparent from the data and discussions within the papers cited:

1. There is always some phosphorus in drainage waters.
2. Total phosphorus levels are always higher than water soluble P in runoff waters -- often substantially higher.
3. Much of the phosphorus in runoff waters is adsorbed on eroded sediments. These sediments may act as a continual source of soluble phosphorus to waters in which they reside.
4. Applications of fertilizer P often, but not always, results in some

Table 2. Phosphorus associated with irrigation and drainage waters.

Researcher(s)	Phosphorus Value(s)	Additional Information
Bolton et al. (1970)	0.12 kg/ha	P loss in tile effluent from unfertilized plots.
	0.19 kg/ha	P loss from fertilized plots; 336 kg/ha, 5-20-10 applied.
Bondurant (1971)	1.98 lb/ac (2.22 kg/ha)	P as ortho-P lost in surface runoff during an irrigation season.
	17.4%	Percent of total outgoing relative to incoming ortho-P (note very low).
Carter et al. (1971)	0.012 ppm	Mean concentration of water-soluble P from 20 subsurface drains; supply water had 0.066 ppm.
	0.066 ppm	Mean concentration of water-soluble P from 5 runoff sites.
Carter et al. (1974)	48.9 ug/1	Mean of water-soluble P values given for 6 surface drains; samples were filtered.
	310 ug/1	Mean of total P values given for 6 surface drains; samples were unfiltered.
Fitzsimmons et al. (1971)	0.05 ppm	Water-soluble P in headwater.
	0.18 ppm	Water-soluble P in surface runoff.
	0.34 ppm	Water-soluble P in open drains.
	0.28 ppm	Water-soluble P in ground water.
	0.32 ppm	Total P in headwater.
	2.39 ppm	Total P in surface runoff.

Table 2. (Continued)

Researcher(s)	Phosphorus Value(s)	Additional Information
Fitzsimmons *et al.* (1971)	0.67 ppm	Total P in open drains.
	0.58 ppm	Total P in ground water.
Graveland (1972)	0.30 mg/l	Mean value of ortho-P in water supply.
	0.24 mg/l	Mean value some distance downstream.
Johnson *et al.* (1965)	0.079 ppm	Flow-weighted average concentration in tile drainage. A high P value was obtained in drainage water from the first irrigation application. P lost was correlated with fertilizer application.
Johnson *et al.* (1976)	1%	Less than 1% of P applied as fertilizer and manure left the watershed in soluble form.
Klausner *et al.* (1974)	0.04 kg/ha/yr	Soluble P concentration in runoff when fertilizer rates were 15 kg/ha and residues were returned.
	0.49 kg/ha/yr	Soluble P concentration in runoff when fertilizer rates were 300 kg/ha and residues were burned. Type of crop had no effect.
Muir *et al.* (1973)	0.09 ppm	Mean concentration of inorganic P found in 4 rivers. P concentrations found in rivers was little correlated with fertilizer use in the surrounding areas.
Olness *et al.* (1975)	1 to 11.5 kg/ha	Total P values. Higher values were found in cropland watersheds and lower values in rangeland watersheds.

Table 2. (Continued)

Researcher(s)	Phosphorus Value(s)	Additional Information
Romkens and Nelson (1974)	0.07 ppm	Mean concentration of soluble P in runoff from a plot with no P fertilizer added.
	0.24 ppm.	Mean concentration of soluble P in runoff when 56 kg/ha P was added in fertilizer form.
Sylvester and Seabloom (1963)	3.2 times	Concentration of soluble phosphate in subsurface drainage water was 3.2 times that in applied water.
	1.12 kg/ha (1 lb/ac)	Soluble P contribution from irrigated areas to the Yakima River.
Thomas <u>et</u> <u>al.</u> (1968)	0.045 kg/ha (0.04 lb/ac)	Available P concentration in runoff from the area where maximum fertilizer rate (40 kg P/ha; 36.5 lb/ac) was applied. P losses showed no definite relationship to crop type or fertilizer rate, but rather was proportional to the amount of soil eroded.
Zwerman <u>et</u> <u>al.</u> (1972)	0.010 mg/l	P concentration in tile drain effluent when fertilizer P was added at a rate of 4.9 kg/ha.
	0.004 mg/l	P concentration in tile drain effluent when fertilizer P was added at a rate of 24.2 kg/ha.

P movement through the soil to tile drains and an increase in soluble P moving to surface drains.

5. Water exports from an irrigated area will have higher phosphorus concentrations than imported water if the incoming water is low in P.
6. On a phosphorus balance basis, often more phosphorus will enter an irrigated area via supply water than will be exported from the area via drainage.
7. Only a small percentage of applied phosphorus will leave an agricultural watershed.
8. P loss from an entire watershed is rarely correlated with fertilizer application. It would appear that losses are better correlated with the amount of soil eroded.
9. Although enrichment ratios of eroded materials may be higher in fertilized areas compared to those unfertilized, fertilizer P applications reduce amounts of P lost via erosion. This is a result of a more vigorous, denser crop on the fertilized field.
10. Management practices of returning residues and applying adequate, yet not excessive rates of P fertilizer will reduce P losses.

If phosphorus is strongly adsorbed by soil then why do any losses occur? The greatest losses of phosphorus to drainage water, as seen in Table 2, are a result of surface runoff. Considering the case presented by Bondurant (1971) one could possibly conclude the following: nutrients in surface runoff from an irrigated watershed will be proportional to two factors -- (1) the amount of erosion in the watershed, and (2) the nutrient content of the eroded material. Bondurant (1971), using Einstein's equation to calculate the average displacement of ions by Brownian movement, makes the point that the velocity of diffusion of ions from the soil to surface applied water is counteracted by infiltration velocities. Ion movement to the irrigation stream by diffusion is therefore minimal and erosion is cited as the cause.

Logically, the more phosphorus in the eroded material, the higher will be the particulate and dissolved P concentrations in the drainage water (see Table 2 -- Romkens and Nelson, 1974). If particles come from a surface soil high in phosphorus, they will tend to support a relatively high concentration of phosphorus in solution; on the other hand, particles low in phosphorus can remove P from solution (Biggar and Corey, 1969; Wildung et al., 1977). The eroded material and dissolved orthophosphate-P may eventually be deposited in a lake, where the soluble P acts as an immediate P source, and the sediment held P acts as a potentially constant source of nutrient P for primary production (Wildung et al., 1974; Latterell et al., 1971).

Leaching losses of phosphorus also occur, but in most cases account for only a small percentage of applied P. Lund et al. (1976) found phosphorus enrichment of soil to a depth of four meters. No percentage lost was given, but concentrations at this depth were low. In Kardos and Hooks' study (1976), each leaching loss was estimated as <3 percent of the total applied. Johnson et al. (1976) found that 1 percent of soluble applied P left the watershed in dissolved (leachable) form. Orth (1976) found concentrations of 0.01 to 0.03 ppm orthophosphate-P to be typical of ground water below an agricultural area in Dade County, Florida. By averaging 63 of 65 reported values gathere in a data survey, White et al. (1963) determined a mean value of 0.011 mg/l of phosphorus in ground waters. Phosphorus is obviously not readily leached!

In summary, the process of erosion -- not leaching -- supplies a substantial amount of phosphorus from agricultural lands to biotic environments.

(c) Nitrogen

"Nitrogen and phosphorus can still be considered as two of the major elements governing primary productivity. In some tropical and highly eutrophic temperate lakes, nitrogen may be a more important limiting factor than phosphorus" (Lund, 1965). Nitrogen's role in eutrophication, then, is mainly the enhancement of eutrophic conditions; permitting growths to develop past an undesirable stage to an objectionable level. The problems associated with eutrophication, as previously discussed, are manifest in extreme eutrophic conditions. Nitrogen, a macronutrient, is required in relatively large amounts for protein synthesis; in the case of algae, nitrogen comprises about 3 to 9 percent of the organisms' dry weights (Mackenthun and Ingram, 1967). A decrease in N supply to lakes, therefore, would have an ameliorating effect.

What nitrogen forms are of concern? Several forms of nitrogen may be utilized as a nitrogen source. The common forms of marine algae attain comparable rates of growth with ammonia, nitrate, and nitrite (Mackenthun and Ingram, 1967). Dugdale and Dugdale (1965) found that the minimum levels of NH_4^+-N (which they equated with NH_3-N), NO_3^--N, and NO_2^--N occurred simultaneously in a lake; maximum levels of the three nutrient ions also occurred simultaneously. This, they surmised, indicated biological utilization of all three. They also determined that maximum assimilation of the element corresponded with minimum circumjacent concentrations of these forms; that is, nitrogen concentrations in the lake were inversely proportional to assimilation. NO_2^--N always occurred in minimal amounts with respect to NO_3-N and NH_4-N, while NH_4-N was always most abundant.

Some 40 species of algae can fix their own nitrogen from N_2 derived from the atmosphere (Garman, 1973). This is an important but non-agricultural source of N for algal nutrition.

What are the agricultural sources of this nutrient element? Viets (1971) states that within the preceding 29 years, fertilizer N use increased greater than 18-fold, and that increases of 3 to 5 percent per year would follow. In 1965, 3.9×10^6 t (4.3×10^6 short tons) of fertilizer N was used in the United States (Smith, 1967) — a vast potential for pollution. In a table of nutrient sources for natural waters (sources included domestic waste, industrial waste, runoff from agricultural land, runoff from non-agricultural land, farm wastes, urban runoff, and rainfall), an estimated 0.68×10^6 to 6.8×10^6 t (0.75×10^6 to 7.5×10^6 short tons) per year of N were attributable to agricultural lands -- the most of any source (Task Group 2610-P Report, 1967)! Agriculture not only has the potential to supply eutrophic nitrogen levels, but apparently is supplying about 60 percent of the nitrogen in United States water supplies (Task Group 2610-P Report, 1967).

Evaluating the role of agriculture (and of particular interest, irrigated lands) in supplying nitrogen to water bodies requires a knowledge of N forms, N dynamics in the soil, and mechanisms of loss from agricultural lands. Mineral forms of N in the soil include nitrate (NO_3^-), ammonium (NH_4^+), nitrite (NO_2^-), nitrous oxide (NO), nitrogen dioxide (NO_2), and elemental nitrogen (N_2) (Stevenson and

Wagner, 1967). Other forms of inorganic N exist in the soil such as intermediates in biological transformations, but these have only transitory residence.

Not all N forms in soil may be transported to aquatic sinks in significant amounts, and not all are produced in significant amounts under "normal" soil conditions. N_2, N_2O, NO, and NO_2 are gaseous forms of nitrogen evolved in soils by the process of denitrification (Allison, 1966) and chemical nitrite decomposition (Bremner and Nelson, 1968); and as such are not readily subject to water transport. Also, they are produced in significant amounts only under the following conditions: N_2O and NO production require low pH -- 6.0 to 6.5 or less; N_2 dominates the denitrification products at pH's above 6; all denitrification requires a supply of readily oxidizable organic compounds, high nitrate levels, and poor drainage (Alexander, 1961). Soils in general, therefore, contain only minimal amounts of these gaseous compounds.

Minimal occurrence of NO_2^- is also a fact, and is explained by Broadbent and Clark (1965) as follows: "pH-values near the upper limit of the range for nitrification sometimes cause an accumulation of nitrite. Under average soil conditions nitrite seldom if ever accumulates. This is because the oxidation of nitrite to nitrate by Nitrobacter generally proceeds faster than the formation of nitrite from ammonium by Nitrosomonas."

From the previously discussed information, one could possibly conclude that most of the inorganic nitrogen in the soil -- 5 to 10 ppm in bare fallow soils in the winter, and 40 to 60 ppm in fertile topsoils in spring and summer (reported by Harmsen and Kolenbrander,

1965) -- is in the form of NH_4^+ or NO_3^-. Cameron (1969) reported values equivalent to 10 to 15 ppm NO_3-N in fallowed soils from most of southern Alberta. Fertilizer application which typically adds in the order of 150 to 600 ppm N to the soil solution per year is also in the form of NH_4^+, NO_3^-, NH_2CONH_2 (which is rapidly hydrolyzed to NH_4^+ according to Harmasen and Kolenbrander, 1965), or NH_3 (which forms NH_4OH in water according to Jenny et al., 1945). Ammonium and nitrate are the predominant inorganic N forms and therefore invite further scrutiny.

The following properties of ammonium help elucidate the mechanisms for its loss from irrigated lands:

1. Ammonium (NH_4^+) is a monovalent cation which is readily adsorbed by soil colloids even at high soil moisture levels (Jenny et al., 1945).
2. Once adsorbed NH_4^+ is not readily exchanged, even by calcium (Vanselow, 1932). This may be explained by the findings of Stanford and Pierre (1946), that NH_4^+ is fixed by soils in identical fashion to K^+, that is, as structural interlayer ions.
3. Ammonium exchanges adsorbed calcium from colloidal surfaces more readily than lithium, sodium, potassium, magnesium, and rubidium (Kelly, 1948), so is said to have high replacing power.
4. Movement ~~of NH_4^+~~ is inversely proportional to fineness of texture (Jenny et al., 1945; Nelson, 1953; Preul and Schroepfer, 1968; Ray et al., 1957).
5. Adsorption of ammonia follows a Freudlich isotherm, that is, the more NH_4^+ added, the less the percentage of added ammonium adsorbed (Preul and Schroepfer, 1968).

In summary, except in cases where high rates of ammonium fertilizers are added to sandy soils (actually soils having low cation exchange capacity), the applied ammonium will be held lightly in or on colloidal particles. Losses will be incurred mainly through the process of erosion.

Although much of the applied N is in the ammonium form, the biochemical oxidation of ammonium to nitrate (a process termed nitrification) results in a large portion of soil NH_4^+ being transformed to NO_3^-. Jahn's work (1971) on twelve Alberta soils exemplifies the extent of the nitrification process. His results indicate that 72 to 88 percent of 50 ppm added NH_4^+ had been nitrified in Chernozemic soils during a 35-day incubation period. He also observed that "with few exceptions the samples contained less than 1 ppm of exchangeable ammonium which indicated that nitrification of exchangeable ammonium had proceeded to completion." These results would support the conclusion that in most cultivated soils the concentration of ammonium is insignificant compared with the quantity of nitrate, as pointed out by Harmsen and Kolenbrander (1965). Optimum moisture and pH conditions for nitrification (Dubey, 1968; Frederick, 1957, respectively) are typical of the study areas. NO_3-N would be expected to be the predominant nitrogen form; little NH_4-N would be expected in the study soils.

The fact that NO_3-N is the predominant inorganic form in soil is not the only basis for Stevenson and Wagner (1970) stating that "nitrate. . . is the form of N which is of greatest concern from the standpoint of pollution of water supplies." If nitrate would stay in the soil there would be little problem indeed. However, as noted by the previously mentioned authors, nitrate is "free to move with the soil water" out of the root zone to the ground water system.

This freedom to move is a result of two properties of nitrate: its electro-negativity and its formation of very soluble salts in the soil. Because of a predominance of negative electrostatic charge within exchange complexes in a soil, NO_3-N, which is also electro-negative, is not adsorbed to any significant extent in soils. The nitrate ion, therefore, will move with water through the soil profile (Gardner, 1965). A "classic" illustration of nitrate movement is given by Krantz _et al._ (1944) in a study involving N movement in a silt loam soil: ". . . a four-tenths inch rain interrupted the soil sampling. The plots were sampled again immediately after the rain, and the data showed that the nitrate content in the surface one-quarter inch, which [illegible] ppm before the shower, dropped to 2 ppm after the shower."

Nitrate salts formed in the soil are highly soluble as indicated in Table 3. Where soil water content remains high, such as in irrigated soils, nitrate will persist in ionic form. When rainfall or irrigation water is supplied in excess of plant use, nitrate may be leached below the root zone (Stanford, 1969). The main mechanism of NO_3-N loss from irrigated lands, therefore, would be leaching.

What can be expected in terms of nitrogen loss from agricultural areas? Table 4 gives several examples. The following conclusions are derived from the data and discussion within the papers:

1. Nitrate moves readily in soils.
2. Movement of nitrate in soil will correspond to water movement in soil. For leaching of nitrates to occur, water must be supplied over and above crop needs and soil retentive capacities.
3. Nitrate concentrations in subsurface drainage waters from fertilized fields are, in general, substantially higher than those from

Table 3. Solubilities of some inorganic nitrate salts.

Compound	Formula	Solubility* g per 100 ml	Temperature* ^{o}C
Calcium nitrate	$Ca(NO_3)_2$	121.2	18
Potassium nitrate	KNO_3	247	100
Sodium nitrate	$NaNO_3$	92.1	25

* Weast (1974).

Table 4. Nitrogen* associated with irrigation and drainage waters.

Researcher(s)	Nitrogen Value(s)	Fertilizer Application	Additional Information
Bolton *et al.* (1970)	4.4 kg/ha/yr	None	Mean discharge in tile effluents from several unfertilized crops.
	8.1 kg/ha/yr	336 kg/ha/yr 5-20-10	Mean discharge in tile effluents from several fertilized crops.
	2 to 3 times	336 kg/ha/yr 5-20-10 plus 112 kg/ha/yr N	Discharge from heavily fertilized corn field relative to other crops receiving less fertilizer.
Bower and Wilcox (1969)	Essentially constant	35 to 100 fold from 1934 to 1963	Though N fertilizer use increased dramatically, no increase in concentrations in the Rio Grande was noticed for the period 1934 to 1963.
Carter *et al.* (1971)	33 kg/ha		Net output from 82,030 ha.
Fitzsimmons *et al.* (1971)	1.04 ppm		Mean concentration in headwaters.
	1.21 ppm		Mean concentration in surface runoff.
	3.19 ppm		Mean concentration in open drains.
	4.92 ppm		Mean concentration in ground water.
	2.02 ppm		Mean concentration of NH_4-N in surface runoff.
Gilliam *et al.* (1974)	1-5 ppm	120 kg/ha N generally	Typical concentration in 60 wells. Several values 10 to 20 ppm were found.

Table 4. (Continued)

Researcher(s)	Nitrogen Value(s)	Fertilizer Application	Additional Information
Graveland (1972)	1.05 mg/l		Mean concentration in supply water.
	0.78 mg/l		Mean concentration some distance [illegible] stream.
Hill and McCague (1974)	6 mg/l and 2 mg/l		Upstream concentration in stream, and concentration 0.4 km (1/4 mi) downstream, respectively. (Extrapolated from two graphs.)
Johnson _et_ _al._ (1965)	2.0 to 14.3 ppm	None	Range of concentrations found in the drainage effluent from below alfalfa.
Krause and Batsch (1968)	88%	112 kg N/ha as NH_4NO_3	Amount of added fertilizer leached as NO_3-N to a 30 cm depth in a sandy soil.
	20 ppm	112 kg N/ha as NH_4NO_3	Minimum concentration in leachate.
	4 to 8 ppm	None	Range of concentrations in leachate from unfertilized soil.
Klausner _et_ _al._ (1974)	0.14 to 1.30 kg/ha	17 to 300 kg N/ha	NH_4-N concentrations in surface runoff.
	0.39 to 29.23 kg/ha	17 to 300 kg N/ha	NO_3-N concentrations in surface runoff. The higher levels where detected when crop residues were not returned and high rates of fertilizer were applied.

Table 4. (Continued)

Researcher(s)	Nitrogen Value(s)	Fertilizer Application	Additional Information
Muir et al. (1973)	1.25 times	168 kg/ha average	Average increase in concentrations found in Nebraska ground water.
	4 times		Increase in fertilizer N use over the same 10 year period.
Nelson (1953)	56 cm (22 in)	672 kg N/ha (600 lb N/ac) as NH_4NO_3	Depth to which added N was leached by 61 cm (24 in) of water applied "judiciously."
Nightingale (1972)	4.4 mg/l		Mean concentration in ground water under 86,500 ha of agricultural land.
	90%		Proportion of the time that NO_3-N was less than 10 mg/l.
Olness et al. (1975)	0.1-20 ppm		The authors found this range of concentrations in surface runoff reported in 23 papers.
	1.0-5.0 kg/ha		Their own values; this range is typical.
Power (1968)	43%		Percent of applied N found in tops.
	27%		Percent of applied N found in roots.
	7%		Percent of applied N found in soluble pool.
	23%		Percent of applied N lost.

Table 4. (Continued)

Researcher(s)	Nitrogen Value(s)	Fertilizer Application	Additional Information
Sommerfeldt and Smith (1973)	19.3%	336 kg N/ha	Proportion of applied N leached to drains (1.52 m deep) over-and-above check losses.
	2.4 and 61.1 kg/ha	0 and 336 kg N/ha, respectively	Loss to drains from check and fertilized plots respectively; first irrigation -- 13 cm of water.
	1.8 and 8.2 kg/ha	0 and 336 kg N/ha, respectively	Loss to drains from check and fertilized plots respectively; second irrigation -- 13 cm.
Sylvester and Seabloom (1963)	37 kg/ha (33 lb/ac)		N exported in irrigation return flow to the Yakima River.
	10 times		Concentration in subsurface drainage compared to applied water.
Thomas and Barfield (1974)	15 ppm		Concentration in tile effluent.
	3 ppm		Concentration in tile effluent plus non-tile drainage waters.

* Nitrogen as NO_3-N unless otherwise stated.

unfertilized fields.

4. More nitrate nitrogen will be lost during the initial irrigation period than in subsequent ones; especially if fertilizer has been applied at high rates.

5. Nitrate concentration in moving water decrease in some instances. This may result in an effectual decrease in agriculture's contribution of nitrates.

6. Plants can remove a large amount of applied N from the soil system; less will be available for leaching.

7. Ammonium may or may not contribute substantially to the nitrogen budget of runoff water from irrigated fields. This depends on what form of nitrogen is applied, whether the applied fertilizer is incorporated before irrigation, and whether the NH_4^+ has been nitrified before irrigation.

8. Levels of nitrate in ground water and surface water on a regional basis are not increased substantially by substantial increases in fertilizer use. Locally, high concentrations of NO_3-N in ground water may occur.

9. Nitrate concentrations in surface runoff, when compared to concentrations in drainage water, are low.

10. Nitrate losses in irrigation return flow are substantial with respect to both eutrophication and farm economics.

11. A large portion of applied N can be lost through poor farm management. Nitrogen can be conserved through judicious use of water and fertilizer.

(c) Carbon

As previously mentioned, algae require at least one hundred times more carbon than phosphorus and about fifteen times more carbon than nitrogen. What is the source of these large quantities of carbon? Carbon compounds entering a water system include CO_2 which has diffused across the air-water interface; dissolved CO_2 in rainwater; H_2CO_3 (carbonic acid), HCO_3^- (biocarbonate), and CO_3^{--} (carbonate) in drainage waters; and organic compounds in waste waters (Ruttner, 1952; Keuntzel, 1969). Keuntzel (1969) points out that "the available free CO_2 from natural inorganic sources (at pH of 7.5 to 9) probably never exceeds 1 mg/l, and it becomes available at a rather slow rate." He suggests that these inorganic sources are not of major importance. Instead he cites the mutualistic symbiotic association of bacteria and algae (in which bacteria give off CO_2 as they metabolize organic materials from municipal wastes, industrial effluents, and dead algae, etc.) as the main source of CO_2 for algal growth. Thus, through the control of organic discharges, algal growth may be controlled.

Goldman et al. (1974) take exception to Keuntzel's literature search and demonstrate by both theory and experiment that the "CO_2-H_2CO_3-HCO_3-CO_3^{--}" buffer system in water can supply CO_2 in excess of algal requirements. CO_2 is released by the carbonate buffer system through the following equilibrium reactions (Goldman et al., 1974):

$$2HCO_3^- \rightleftharpoons CO_3^{--} + H_2O + CO_2 \qquad (1)$$

$$HCO_3^- \rightleftharpoons CO_2 + OH^- \qquad (2)$$

$$CO_3^{--} + H_2O \rightleftharpoons CO_2 + 2OH^- \qquad (3)$$

Under typical pH levels (8 to 8.5) the bicarbonate ion is the dominant carbon species; equation 2 represents the process by which CO_2 is supplied for primary production. According to Goldman et al. (1974) this reaction can release CO_2 at a rate in excess of eight times algal comsumption.

Natural carbon sources usually meet minimal requirements for algal nutrition (Greeson, 1969). Perhaps this explains why very little information is available on the role of irrigation return flow in supplying C to aquatic environments -- it is not important.

Sediment as a Physical Pollutant

A. Problems

In a preceding section on phosphorus, sediment was mentioned as a storehouse of nutrient P for eutrophication. This role of sediment in water pollution is only one of many as the following list of effects of sediments demonstrates:

1. Impact on municipal use. The tolerable-turbidity[1] level for drinking water is 5 mg/l (McGauhey, 1968). Suspended loads giving higher turbidity values than this are visibly detectable in a glass of water. Unfortunately, public preference for clear water is costly since the required, chemically-enhanced, flocculation to remove suspended materials is expensive.
2. Impact on industrial use. Similarly to municipalities, some industries must pretreat their water supply. For example, canning requires turbidities $\leq$10 mg/l; textile industries $\leq$5 mg/l; and water

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[1] One indirect measure of suspended load is the measurement of turbidity. Suspended sediment values are usually lower than turbidity values (Stall, 1972).

for cooling should have turbidity levels $\leq$50 mg/l (McGauhey, 1968).

3. Impact on reservoirs. The storage capacity of reservoirs is being gradually reduced by sedimentation -- 0.2 percent per year on the average (Spraberry, 1965).
4. Impact on natural or constructed channels. According to Stall (1972) rivers must carry sediment in order to maintain an inherent energy balance. Robinson (1971) states that a stream will attempt to erode its banks or bed to obtain this sediment. Thus, a lack of sediment can result in instability in water channels. Deposition of sediments will occur if the suspended material exceeds the stream's inherent ability to carry the sediment load. Thus, an excess of sediment can reduce the efficiency of man-made water channels through resultant deposition.
5. Impact on fish. Buck (1956) has shown that fish production in ponds stocked with Large Mouth Bass, Bluegills, and Redar Sunfish is inversely proportional to sediment concentrations. At a level of 25 ppm suspended sediment, production was 180.9 kg/ha, while in ponds having >100 ppm production dropped to 32.8 kg/ha. Spawning was affected by sediment concentrations of 84 ppm in the water.
6. Impact on recreation. "Swimming, boating, and water skiing are forms of water-based recreation that are directly affected by the turbidity and the amount of sediment in the water. The authors have no way of expressing this adverse effect of sediment, or the beauty of a clear stream, except to call attention to the obvious" (Glymph and Storey, 1967). Oschwald (1972) states that complaints are heard from swimmers and boaters when turbidity levels reach about 10 and 20 mg/l, respectively.

B. Sediment sources

There are 3.6 billion tons of sediment washed into tributaries in the United States per year (Wadleigh, 1968). In colloquial jargon, "Where does all that come from?!" According to Wadleigh (1968) approximately half of this sediment has been eroded from agricultural land. Sheet erosion, rill erosion, and gully erosion by wind and by water are all contributors.

In assessment of the role of irrigation agriculture in supplying sediments to natural water systems, an examination of the definition of erosion will be of benefit. Erosion defined simply is the detachment and transport of soil material (Johnson and Molderhauer, 1970). In a sprinkle-irrigated area, water is applied at rates so as not to exceed the infiltration capacity of a soil. Thus, there is a means of soil particle detachment but no means for transport. Little soil erosion by irrigation water in such an area would be expected. In a flood-irrigated area, however, water application rates must exceed infiltration rates to facilitate water distribution. As the applied stream flows over the field there is a means for both detachment and transport of soil particles. Substantial soil losses by erosion are possible in flood-irrigated areas.

Table 5 gives a listing from the scant literature data available on sediment in irrigation supply and drainage waters. The following conclusions are derived from the reported data and from discussion within the papers:

1. The soil may act as a filter by retaining the sediment in applied irrigation water.

Table 5. Some reported sediment values for waters associated with irrigation.

Researcher(s)	Sediment Value(s)	Additional Information
Bondurant (1971)	48.5 ppm	Concentration in applied water, 1966.
	30.3 ppm	Concentration in surface runoff, 1966.
	26.2 ppm	Concentration in applied water, 1967.
	42.3 ppm	Concentration in surface runoff, 1967.
	100.4 ppm	Concentration in applied water, 1968.
	242.3 ppm.	Concentration in surface runoff, 1968.
Brown *et al.* (1974)	4.0 t/ha	Net amount eroded from farms in the Northside tract.
	1.42 t/ha	Net amount eroded from farms in the Twin Falls tract.
	4.5 t/ha	Amount of sediment deposited in the drains of the Northside tract.
	0.95 t/ha	Amount of sediment deposited in the drains of the Twin Falls tract.
Carlile (1972)	91 mg/l	Concentration in applied water.
	1,300 mg/l	Concentration in runoff before a settling pond.
	400 mg/l	Concentration in runoff after a settling pond.
Carter (1971)	2.78 t/0.1233 ha-m (3.06 t/ac-ft)	Net amount eroded from a large irrigation tract in Idaho.

Table 5. (Continued)

Researcher(s)	Sediment Value(s)	Additional Information
Fitzsimmons *et al.* (1971)	195 ppm	Concentration in headwaters.
	1,549 ppm	Concentration in surface runoff.
	586 ppm	Concentration in open drains.
Graveland (1972)	14.6	Turbidity units at Bassano Dam.
	7.0	Turbidity units at inlet of Brooks aquaduct.
	5.0	Turbidity units at point of outflow from Lake Newel.
	46.0	Turbidity units at Matzhiwin Spill.
	10.0	Turbidity units at 12 Mile Spill.

2. Runoff water often has substantially higher sediment concentrations than the corresponding applied water. The net change varies from year to year and from place to place.
3. The net amount of sediment exported to a natural body of water will be affected by the amount of deposition in drainage channels.
4. Sedimentation can significantly reduce suspended loads before the drainage water is discharged into a natural sink. Natural or constructed sedimentation basins are helpful in reducing sediment loads, but further treatment may still be required to reduce turbidity.
5. The concentration of sediment in supply water can vary greatly depending on the extent of erosion of channel banks, the settling time and trap efficiency of storage structures, and the amount of natural runoff received by the irrigation network.
6. Furrow irrigation gives higher sediment yields than other methods of water application.
7. There is sufficient erosion in irrigated areas to be of both economic and environmental concern.

Dissolved Salts

A. Problems

To discuss all the problems associated with specific and total dissolved salts would be beyond the scope of this work. For more detailed information the reader is invited to refer to the following publications: National Technical Advisory Committee (1968); American Society Testing Materials (1967); and/or Utah State University Foundation

(1969). Only a brief discussion of factors important in municipal, industrial, ecological, recreational, and animal use of water will be given here. More attention is paid to the problems associated with dissolved salts as they affect irrigation agriculture.

Dissolved solids are not removed to any extent by conventional water treatment (Utah State University Foundation, 1969). Municipalities and industries must therefore locate themselves in proximity to water of inherent high quality. Any factors, such as irrigation return flow, which may jeopardize that quality, impose a threat to the public and to private investors. The following is a list of problems associated with polluted municipal water supplies as affected by specific and total salts as given by the Utah State University Foundation (1969): (1) objectionable physiological effects and taste as a result of high total dissolved solids; (2) cost of soap and water softening as a result of hardness; (3) objectionable taste of chloride in water; (4) objectionable taste and laxative proper f sulfate; (5) harmful effect of sodium on persons having cardi al, or circulatory diseases; (6) possibility of methemoglobinemia (nitrate poisoning of infants, to be discussed later); (7) various problems associated with eutrophication of domestic water supplies; and (8) adverse effect of phosphates on flocculation during water treatment.

Industrial uses of water are numerous and the corresponding potential problems are also manifold. Because of variations in quality requirements -- for example, the manufacture of television tubes requires water containing dissolved solid concentrations of only parts per billion (Partridge, 1967), while waters having up to 35,000 parts per million have been used for cooling purposes (National Technical Advisory

Committee, 1968), total dissolved salts may be a problem for one industry but not for another. A broad generalization can be made, however, that the lower the dissolved solids the fewer the number of dollars that must be expended by industry for water treatment.

When considering the effect of dissolved solids on aquatic flora and fauna, an ecological approach should be taken. Naturally occurring fresh water organisms inhabit the environments to which they are adapted. The inter-dependence of organisms is complex, and an upset in species composition may set off a chain reaction. Such a change in biota may be a result of fluctuations in the chemical environment. Total dissolved solids are not a major concern in considering these changes because they are not prone to large or rapid natural fluctuations. However, if the dissolved solids composition of receiving waters is increased by more than one-third the natural level or raised in some way to exceed an osmotic pressure equivalent to 1,500 mg/l NaCl (National Technical Advisory Committee, 1968), such as by salts discharged in irrigation return flow, then an upset in the biotic community can be expected.

With respect to water recreation and aesthetics, changes in dissolved solids are undesirable only to the extent that they adversely affect domestic consumption and aquatic ecosystems.

Animals will tolerate much higher salt concentrations than will humans (McGauhey, 1968). McGauhey (1968) cites McKee and Wolf (1963) in giving 2,500 mg/l as a threshold value for total dissolved solids "at which sensitive animals show slight effects from prolonged use of such water. Lower concentrations are of little or no concern."

Both total salinity and specific ion concentrations of water supplies are characteristics that concern the irrigation specialist. Reeve and Fireman (1967) estimate that at least one-third or more of the world's irrigated lands suffer from problems caused by salts. The upstream cycle of diversion, distribution, application, runoff, percolation, and reuse can seriously modify the water quality for the downstream user. For example, Thorne and Thorne (1951) found a 20-fold increase in salts along a 322 km (200 mi) stretch of a Utah river from which irrigation water was diverted seven times. The minimum salinity increase in irrigation return flows, as derived by McGauhey (1968) in his review of seven studies in the western states, was five times the level of the salt concentration in the corresponding supply water. The irrigation return flow cycle is a salt donor and salt concentrator. Reuse of this water can increase soil salinity problems already prevalent in many areas, and can enlarge the area of salt-affected soils.

Excess salinity delays or prevents seed germination and reduces the amount and rate of plant growth. These problems are associated primarily with high osmotic pressures which tend to induce plasmolysis. For each increase in salt content equivalent to an electrical conductivity of 4 mmhos/cm, the plant must overcome an additional force equivalent to 1.44 atmospheres of pressure to obtain its water (United States Salinity Laboratory, 1954). Some problems may be due to nutritional imbalance or toxicity caused by specific ions. Bernstein (1961) relates several effects of salts on plant growth: (1) plant growth is inversely proportional to osmotic pressure of nutrient solutions; (2) fruit crops suffer at even low chloride and sodium concentrations, while vegetable and field crops are not specifically sensitive to chloride or sodium ions; (3) boron is toxic at a few ppm; (4) bicarbonate interferes with iron utilization by plants; (5) high sulfate concentrations decrease calcium uptake by lettuce; and (6) calcium may restrict the uptake of potassium in beans and carrots.

B. Sources of salts

Why are dissolved solid concentrations in irrigation return flows higher than those in applied waters? Supplementary water applied to crops acts as a solvent and transport mechanism for nutrients, maintains plant turgidity, and serves as a coolant in the process of transpiration. While the consumptively-used water lost from the soil-plant system is pure, the imported irrigation water contains varying amounts of salt dissolved during intimate contact with geologic material. Only a portion of the applied salt is removed by crops. For example, Pratt and Chapman (1961) found that about one-fourth of the salt that they added in low salinity wat to lysimeters was removed by barley and Sudan grass. The substantial quantity of salt added to the soil in irrigation water is further concentrated in the soil by evapotranspiration (Rhoades and Bernstein, 1971), and is only partially removed by crop growth. Repeated water application and evapotranspiration result in a salt build-up in the soil. Because the maintenance of viable irrigated lands requires the removal of accumulated salts from soil (Bouwer, 1969; Scofield, 1940; Wilcox, 1963), irrigation water is applied at some fraction over-and-above crop needs in order to "wash out" the imported salts. This saline leaching fraction percolates below the root zone into natural or artificial drainage systems and re-enters surface or ground water supplies. Said in other words, the saline leaching water becomes return flow.

Dissolution of salts in soil (Bower, 1974), miscible displacement of soil solutions by irrigation water (Sadler et al., 1965), weathering of soil-silicate minerals (Rhoades et al., 1968), decreased leaching fraction (Rhoades et al., 1973), decreased rate and increased frequency of water application (Keller and Alfaro, 1967), and little dilution from non-percolating water (Bower, 1974) all contribute to an increase in the total dissolved salt concentration of drainage water. Precipitation of salts (Rhoades et al., 1974) and removal of soil solutes by crops (Bower, 1974) are processes which reduce salt loads. However, much less than one-half of the salts can be effectively removed in this manner. Some leaching of salts remains necessary.

Salts in irrigation return flow may also vary qualitatively from applied irrigation water. Of particular interest is the increase of Na^+ relative to Ca^{++} and Mg^{++} since Na^+ plays a major role in the deterioration of soil physical properties. Soils having an exchangeable sodium content in excess of 15 percent of the total cation exchange capacity, that is, an exchangeable sodium percentage (ESP) greater than 15, are termed alkali soils (United States Salinity Laboratory Staff, 1954). Increases in ESP values in soil are indicative of conditions which cause nutritional deficiencies and diminished air and water permeability (Eaton et al., 1967; van Schaik, 1967).

Increased ESP values can result from two processes: (1) the precipitation of calcium and magnesium salts as their solubility limits are exceeded; (Sodium salts are highly soluble and therefore remain in solution at "normal" soil concentrations, whereas the solubility limits of magnesium silicates, calcium sulfate, and calcium and magnesium carbonates are often exceeded in soil solutions. This causes an effectual increase in Na^+ concentration. At higher concentration Na^+ replaces more Ca^{++} and Mg^{++} from the exchange complex resulting in higher ESP values.) and (2) the addition of irrigation water having a high sodium to calcium-plus-magnesium ratio, that is say having a high sodium-adsorption-ratio (SAR). (Since adsorbed ions are in "dynamic" equilibrium with the ions in the added irrigation water, the SAR of the imported solution ultimately controls the soil ESP values after prolonged water additions, as explained by Babcock _et al._, 1959.)

The sodium ion concentration in irrigation return flows may be drastically altered as the water passes through geologic strata. Geological deposits of marine origin are high in sodium salts, and depending on the geochemical abundance of these salts, the duration of the contact time of water with salt, and the length of the flow path (Edmunds, 1976), the process of solution can add substantial quantities of Na^+ to the percolating return flow. Sodium concentrations can also be increased by exchange phenomena in which the geologic material acts as a semipermeable membrane. This occurs because calcium and magnesium have a greater affinity for exchange sites than does sodium -- the calcium and magnesium are removed from solution as they replace the adsorbed sodium. Obviously, the sodium concentration in the percolating water increases.

In summary, the processes associated with leaching can result in significant specific and/or total salt additions to irrigation return flows, which, when discharged into a water supply, often hamper beneficial water use.

The data in Table 6 portray the extent of salt increases in irrigation return flow. The following are conclusions derived from the

Table 6, Dissolved salts associated with irrigation and drainage waters.

Researcher(s)	Dissolved Salt Value(s)	Additional Information
Bondurant (1971)	336 mg/l	Mean concentration in applied water, 1966.
	336	Mean concentration in runoff water, 1966.
	246	Mean concentration in applied water, 1967.
	249	Mean concentration in runoff water, 1967.
	259	Mean concentration in applied water, 1968.
	229	Mean concentration in runoff water, 1968.
		(All ppm values were calculated from given electrical conductivity data.)
Carter (1971)	2.4 t/ha	Net total output from a large irrigation tract in Idaho.
	2.25 times	Subsurface drainage waters had total salt concentrations approximately 2.25 times that of irrigation waters.
Eldridge (1963)	3 to 10 times	Irrigation return flows contain at least 3 times and often 10 times as much salt as diverted water according to this author.
Graveland (1972)	1.32 times and 4 to 5 times	Increase in total dissolved salts and sodium, respectively, from the point of diversion to the point of return to receiving rivers.

Table 6. (Continued)

Researcher(s)	Dissolved Salt Value(s)	Additional Information
McGauhey (1968)	7.0 times	Mean increase in salinity values of return flows with respect to irrigation water as derived from 7 reports.
Sylvester and Seabloom (1963)	56%	Proportion of the salt in the Lower Yakima River that was contributed by irrigation return flows.
	38 and 4.1 meq/l; 9.2 times	Concentration of sodium in subsurface drainage water compared to applied water, respectively; and proportional increase of sodium in the drainage water.
Wilcox (1962)	8 times	Increase in dissolved solids over a 724 km (450 mi) stretch of the Rio Grande.
	50 to 75%	"It would appear that irrigation might account for 50 to 75 percent of the increase. . . ."

data:

1. Irrigation return flows can and often do carry vast quantities of dissolved salts to natural receiving waters.
2. Salt concentrations are not substantially increased in surface runoff; leaching and drainage are the processes by which substantial salt loads return to water supplies.
3. The degree of increase in salt loads in returned waters varies from district to district, and river to river.
4. Downstream users will have more extensive salt problems than upstream users.

C. Infantile nitrate poisoning (methemoglobinemia)

Nitrate poisoning of infants can occur at NO_3^- concentrations of 45 mg/l (Utah State University Foundation, 1969). This level is approximately equivalent to 10 ppm NO_3-N. The factors governing the amount of NO_3-N in water supplies have been previously discussed in some detail under the section on eutrophication in this review and will not be repeated here. As witnessed by the data in Table 4, NO_3-N levels in excess of critical levels are not common in irrigation return flows. At a few individual sites, ground water recharge by percolating waters has increased nitrate concentrations past permissible levels. The closer that a water source is to a concentrated nitrate supply, with respect to both space and time, the greater the chance of nitrate problems. Excessively high nitrate concentrations do not appear to be a problem in irrigation return flow except where high rates of nitrate fertilizer application and subsequent leaching are the norm. Judicious use of fertilizer and water can alleviate nitrate problems.

Thermal Pollution

Fish kills can result from rapid fluctuations in water temperatures. Also, various species of fish and their associated biota are adapted to relatively narrow, specific temperature ranges (National Technical Advisory Committee, 1968). Depending on the temperature of the water source, irrigation can either increase or decrease water temperature, that is, the process of irrigation has a temperature-moderating effect on return waters. Few data are available on the effect of irrigation agriculture on this type of pollution.

Pesticides

Some pesticides are resistant to biochemical degradation and as such persist in the environment and may be found in irrigation return flows. An accumulation of these occurs in the food chain in a progressive manner as the ultimate consumer is approached. The accumulation of pesticides in organisms to toxic levels is the point of concern. Other pesticides are rapidly degraded and are of little ecological significance. "The rapid increase in the use of less persistent pesticides such as the organophosphates and carbonate compounds has not led to an increase in their occurrence in the major streams of the United States" (Merkle and Bovey, 1974).

Bailey and Hannum (1967) made a study of the occurrence of 23 pesticides in surface waters, agricultural drainage, sediments, and aquatic organisms in California. In drainage waters from six irrigated areas they found a maximum of 1.867 ug/l, a minimum of 0.010 ug/l, and a mean of 0.402 ug/l for total identified pesticides. In the overall study area, concentrations of pesticides in sediments reached values two orders of magnitude greater than those found in the drainage water, and concentrations in aquatic organisms reached values four orders of magnitude greater. Except for one maximum, all the levels of the monitored pesticides in the drainage waters were lower than the toxicity values for fresh water organisms as given by the National Technical Advisory Committee (1968); but concentrations in sediments exceeded those toxicity values in the majority of cases. Information on the pesticide-supplying power of sediments to water systems would be useful. However, the pesticide levels in the drainage waters indicate this supplying capacity to be minimal.

Values given by the Inland Water Directorate (1975) for 22 pesticides at a point downstream from the return flow study sites are all minute. The study areas do not appear to be significant sources of pesticides.

Water Quantity as it Affects Quality

When considering electrolytes, it is obvious that mixing waters of differing quality will result in a solution which has characteristics intermediate to the two solutions mixed. Thus, water of low quality can be improved by addition of water having higher quality. Thomas and Barfield (1974) demonstrated this to be important in a study of nitrate losses in drainage waters. They found that tile effluents had NO_3-N concentrations of 15 ppm whereas the NO_3-N concentration for total seepage water (tile effluent plus non-tile seepage) was 3 ppm. Bower (1974) points out that the quality of drainage water is also affected by the amount of runoff removed by drains -- runoff water quality is generally much better than that of seepage water.

Quality can also be affected by the total amount of flow in a channel. Increased flow usually results in a decrease in the ratio of water volume to bed-surface area, that is, there is more water flowing in the channel per unit area of water-earth contact. Thus, there is a decrease in potential dissolved solids per unit of water flow. Also, any polluting discharges into a river will not be increased by increased river flow. Instead, the dilution factor will be increased. These are the factors that led the Biology Department, Heidelberg College (1971) to determine whether water quality could be controlled through flow augmentation. While concentrations of compounds, generally speaking, can be controlled in this manner, net exports are increased by flow augmentation.

In terms of dilution it is interesting to note that the total "apparent return flow" from the B.R.I.D. is in the order of 300 ha-cm/hour (294 cfs) as estimated by the data given by Kerber (1975), and the mean discharge of the receiving waters is approximately 7,000 ha-cm/hour (6,867 cfs) as given by the Research Council of Alberta (1970).

Water Quality Criteria

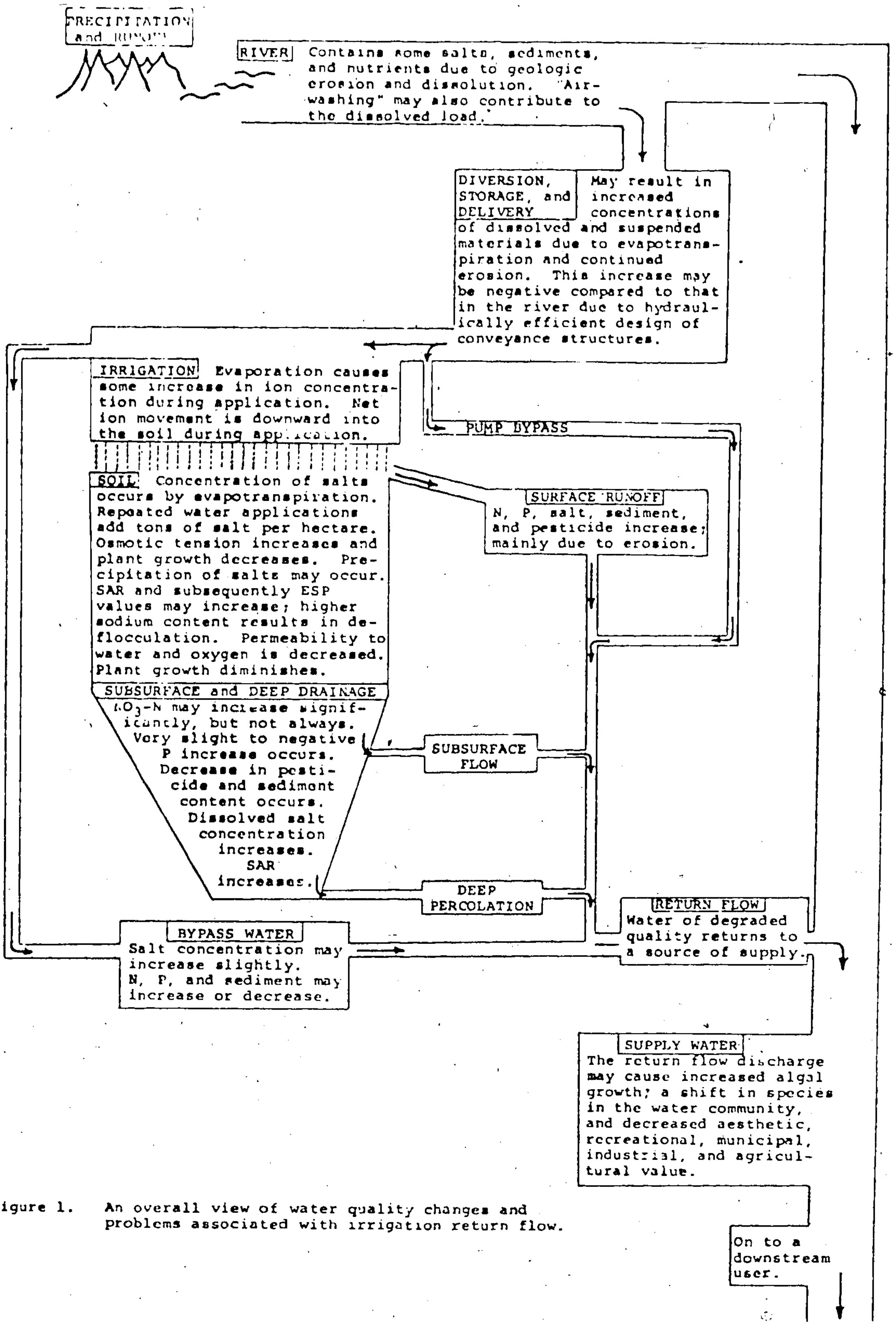
Action to control return flow constituents must be preceded by both a categorical inventory and a comparison of findings to criteria. The criteria must serve as a basis for judgement of pollution potential of the constituent in question. Matters would be simplified if specific criteria applicable to a broad spectrum of uses could be developed, but specific criteria usually suit specific situations. To illustrate the point, the needs of a trout do not correspond one-to-one with the needs of a pre-mix concrete industry. Quality control objectives can be summarized by two generalizations, however: (1) discharged water should not cause rapid, extensive, or extreme fluctuations in water quality typical of natural conditions found in the receiving body; and (2) water quality control must be designed to protect the most sensitive user.

The information given by the National Technical Advisory Committee (1968) and an updated version of the same report by the National Academy of Sciences (1973) are excellent reviews on water quality criteria. From these two publications and from the information given by the United States Salinity Laboratory Staff (1954), the water quality criteria presented in Table 7 were derived. Criteria listed under "industrial" are suitable for the most sensitive industries (food processing) found in the downstream city of Medicine Hat.

There is some danger in using such criteria as standards. The values do not have universal application and are guidelines rather than regulations. They are useful, however, inasmuch as these criteria do form a basis for assessment of quality degradation.

Summary of Literature Review

Figure 1 depicts a general overview of quality changes and problems associated with irrigation return flow. Leaching is the main mechanism that increases the total dissolved salt, nitrate, and sodium content in waters returning from irrigated lands; while erosion, which is mainly attributable to surface runoff, is responsible for the vast



RIVER Contains some salts, sediments, and nutrients due to geologic erosion and dissolution. "Air-washing" may also contribute to the dissolved load.

DIVERSION, STORAGE, and DELIVERY May result in increased concentrations of dissolved and suspended materials due to evapotranspiration and continued erosion. This increase may be negative compared to that in the river due to hydraulically efficient design of conveyance structures.

IRRIGATION Evaporation causes some increase in ion concentration during application. Net ion movement is downward into the soil during application.

SOIL Concentration of salts occurs by evapotranspiration. Repeated water applications add tons of salt per hectare. Osmotic tension increases and plant growth decreases. Precipitation of salts may occur. SAR and subsequently ESP values may increase; higher sodium content results in deflocculation. Permeability to water and oxygen is decreased. Plant growth diminishes.

SURFACE RUNOFF N, P, salt, sediment, and pesticide increase; mainly due to erosion.

SUBSURFACE and DEEP DRAINAGE NO_3-N may increase significantly, but not always. Very slight to negative P increase occurs. Decrease in pesticide and sediment content occurs. Dissolved salt concentration increases. SAR increases.

BYPASS WATER Salt concentration may increase slightly. N, P, and sediment may increase or decrease.

RETURN FLOW Water of degraded quality returns to a source of supply.

SUPPLY WATER The return flow discharge may cause increased algal growth; a shift in species in the water community, and decreased aesthetic, recreational, municipal, industrial, and agricultural value.

On to a downstream user.

Figure 1. An overall view of water quality changes and problems associated with irrigation return flow.

Table 7. Maximum permissible or maximum desirable levels of the studied quality components as related to various uses in southern Alberta.

Quality Component	Use (1) Recreational and Aesthetic	 (2) Aquatic Life	 (3) Municipal	 (4) Industrial	 (5) Irrigation
Total dissolved salts	RP*	1 1/3 times natural level, maximum	<500 mg/1 desirable	500 mg/1, maximum	<500 mg/1 desirable
Hardness	RP	RP	<150 mg/1 desirable	250 mg/1, maximum	RP
Potassium	RP	RP	RP	RP	RP
Sodium	RP	RP	RP	RP	SAR 8-18,plants; <1-2, soils desirable 1.25 meq/1 residual sodium carbonate
Bicarbonate plus carbonate	RP	RP	RP	RP	
Chloride	RP	RP	250 mg/1, maximum	250 mg/1, maximum	35.5-710 mg/1, maxima
Nitrate nitrogen	As affects (2) and (3)	Maintain the natural N to total P ratio	10 mg/1, maximum	10 mg/1, maximum	RP
Phosphorus (total)	As affects (2) and (3)	50 ug/1 in lakes, 100 ug/1 in running waters, maxima	<50 ug/1 to restrict aquatic growth;<100 ug/1 to avoid dispersion of suspended materials	As affects (2) and (3)	RP

Table 7. (Continued)

Quality Component	Use (1) Recreational and Aesthetic	(2) Aquatic Life	(3) Municipal	(4) Industrial	(5) Irrigation
pH**	6.5 - 8.3	7.0 - 9.0 for flora; 6.0 - 9.0 for fauna	6.0 - 8.5	6.5 - 8.5	5.5 - 8.5
Suspended load	Clear waters desirable	25 mg/l for a high level of protection; 80 mg/l for moderate protection	Extremely variable; in some instances higher values are desirable	10 mg/l, maximum	Little data available for assessment
Temperature	RP	Maintain natural daily and seasonal values	$30^{\circ}C$; $\pm 3^{\circ}C$ from ambient, maxima	RP	RP

* RP denotes rarely a problem, i.e., levels seldom exceed critical values.

** Data given are ranges between minimum and maximum values.

majority of sediment, total phosphorus, and pesticide in drainage waters. Beneficial downstream uses including recreational, biological, municipal, industrial, and agricultural activities may be impaired by the quality fluctuations imposed on water supplies by irrigation return flow. Inventory of these quality fluctuations and comparison with desirable criteria are initial steps in water quality management.

MATERIALS AND METHODS

Physiography of the Hays and Vauxhall Study Areas

Location and Size

As depicted in Figures 2 and 3, the study areas are located in the central portion of southern Alberta between the Bow and Oldman Rivers near their confluence to form the South Saskatchewan River. The Hays study area includes 3,010 ha located in Townships 11 and 12, Ranges 13 and 14, west of the 4th Meridian. The Vauxhall area encompasses 1,350 ha in Township 14, Range 16, west of the 4th Meridian.

Climate

All data on climate except wind speed and cloud cover are taken directly, or calculated from the data given by the Committee of the Canada Department of Agriculture (1960). The other data were extrapolated from graphs by Longley (1972).

This portion of the province has a semi-arid climate. With a mean annual precipitation of only 31.2 cm (12.3 in), the average moisture deficiency for this region is 27.2 cm (10.7 in). Precipitation extremes include the values of 19.1 cm (7.5 in) and 61.7 cm (24.3 in), with the majority of years being droughty. The distribution of precipitation is favorable for agriculture since 54 percent falls during the 4-month period of May through August. Average temperature for the year is only 4.2°C (39.5°F) due to a lengthy winter, but the length of the growing season is about 152 days with an average of 3,492 degree-days (6,286 degree-days using the Fahrenheit scale) above 0°C.

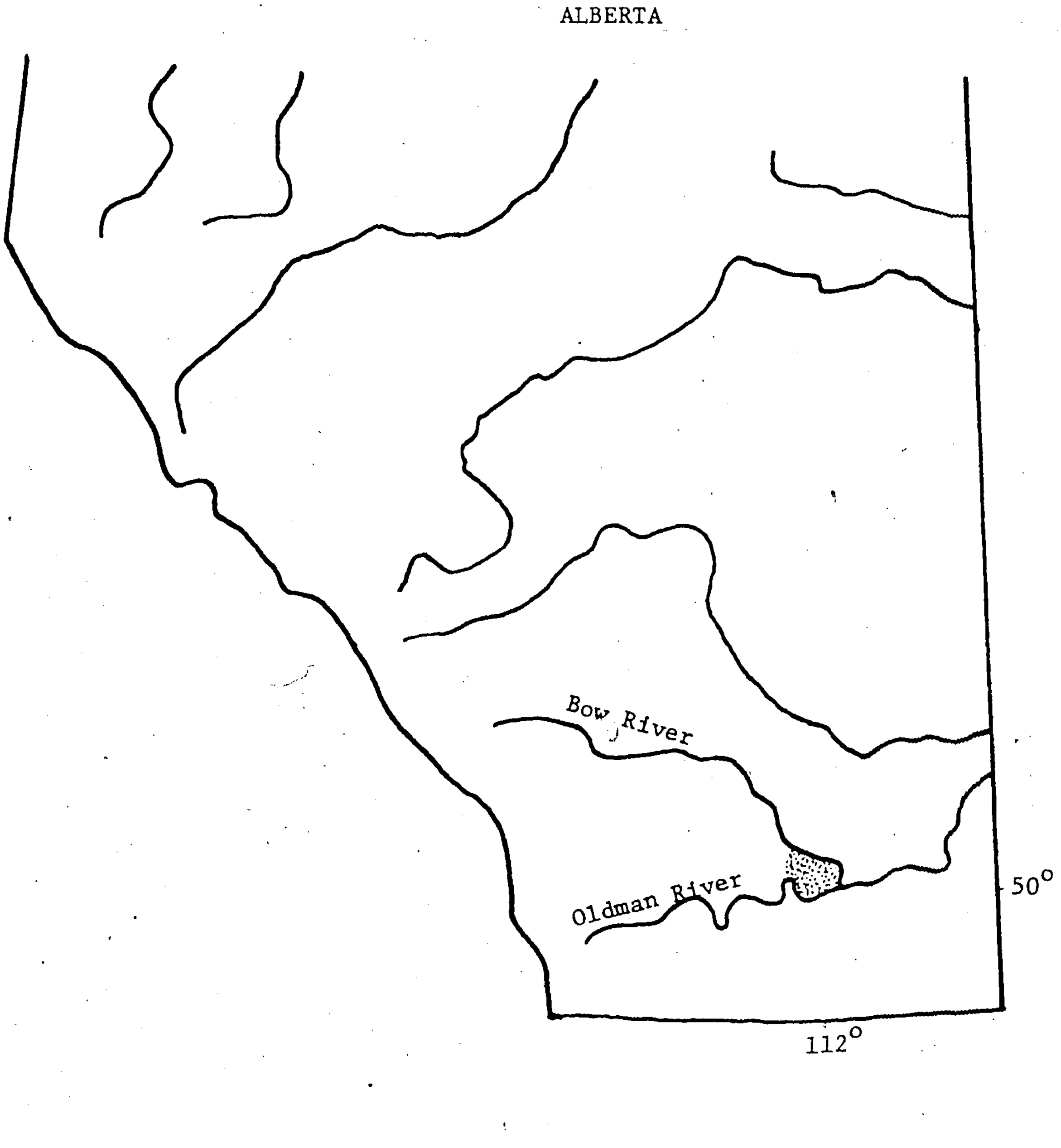


Figure 2. Location of the study areas in southern Alberta.

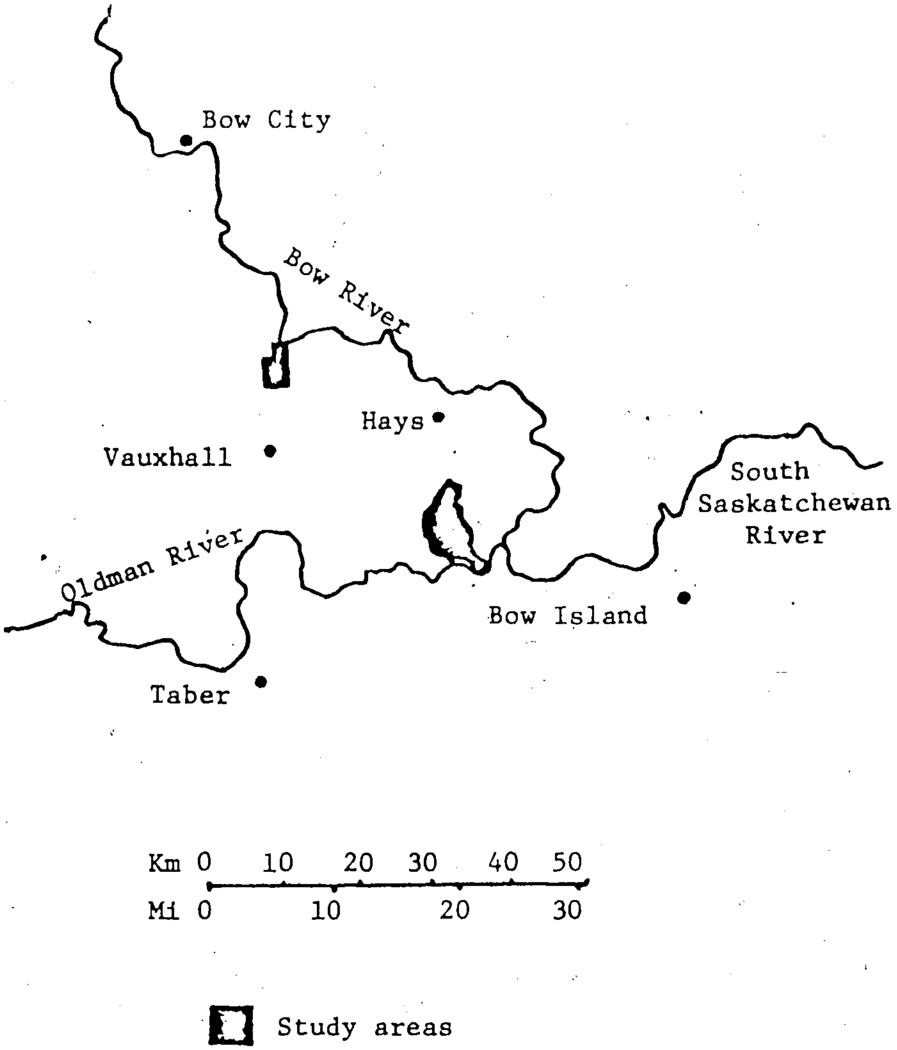


Figure 3. Location of the study areas near the confluence of the Bow and Oldman Rivers.

Mean average temperature for the summer season of June through August is 16.9°C (62.5°F). Wind speed and cloud cover both decrease in the summer months to minimum values in July of 19.3 to 20.9 km/hr (12 to 13 mph) and 4.3 tenths, respectively.

Vegetation

The June grass (Koeleria cristata), blue grama grass (Bouteloua gracilis), common spear grass (Stipa comata), and western wheat grass (Agropyron smithii), dominating the natural vegetation in areas of loamy soil, are interspersed with sage (Artemisia sp.). Common spear grass (Stipa comata) dominates the sandy soil regions with smatterings of sage (Artemisia sp.) and prairie roses (Rosa sp.) adding variety The sighting of various cacti (Opuntia polyacantha and Opuntia neomanilaria) is not uncommon in drier areas (Bowser and Kjearsgaard, 1952).

The vast majority of the landscape in the two areas is covered with cultivated vegetation. Alfalfa, barley, wheat, potatoes, and sugar beets are the dominant crops while some corn, peas, rapeseed, flax, oats, and beans can be found.

Topography

The vast majority of the Hays area has nearly level (0 to 0.5 percent slope) to gently undulating topography (0.5+ to 2.0 percent slope), but the small non-irrigated portion on the eastern edge is undulating (2+ to 5 percent slope). There is a general easterly slope. Except for about 1 km^2, the Vauxhall area is all nearly level to gently undulating with an overall, northerly slope.

Bedrock Geology

Though less than 30 km (18.6 mi) separate the Hays and Vauxhall areas, the tills at these two locations are underlain by different bedrock formations. The Foremost formation consisting of gray and pale brown sandstone, gray and green siltstone, green and gray shale, carbonaceous shale, ironstone and occasional coal seams is found in the Hays area. This formation, which varies in thickness from 45 to 120 m (150 to 400 ft), is non-marine in origin. A younger formation, the Oldman, underlies the Vauxhall area. This formation, ranging in thickness from 90 to 180 m (300 to 600 ft), is also non-marine in origin. It consists of massive, crossbedded medium to course-grained sandstone, gray clayey siltstone, green and gray shale, and ironstone concretionary beds. The Geological Survey of Canada (1971) and Jackson (1975) give the above information in pictoral and verbal form.

Surficial Geology

Bayrock and Jones (1963) and Bowser and Kjearsgaard (1952), respectively, describe the surficial geology of the Vauxhall and Hays study areas. The glacial till deposits in this portion of the province are of Wisconsin glacial age, deposited by a Keewatin glacier which advanced from a northerly direction. In the Hays area, the resultant till layer varies in thickness from 9 to 15 m (30 to 50 ft). There is little variability of the loam to clay loam texture of the till with depth. The water permeability of the till is favorable for irrigation practices and the salt content is low. Post-glacial action has resulted in deposition of aeolian, alluvial, and alluvial-lacustrine materials in depths from 30 cm (1 ft) to 2 m (6 ft) or more.

Glacio-lacustrine material is found over the entire Vauxhall study area except for the ground moraine in the southwestern tip. The ice-marginal lacustrine deposit is more clayey near the surface than at depth. A large percentage of very fine sand and silt may be found below the solum. The underlying till in the Vauxhall region varies from 6 m (20 ft) to 12 m (40 ft) in thickness.

Soils

As depicted in Figures 4 and 5, Orthic Brown Chernozems typically found in the short-grass prairie regions of the province dominate the soils of both the Hays and Vauxhall areas. In the Hays portion, a strip of Regosolic soil borders the Oldman River, but the remainder of the area is Chernozemic. Tongues of soil mapping units, which include two Eluviated Brown Chernozems, a Solodic Brown Chernozem, and a Brown Solonetz in their associations, dip into the Vauxhall area. Again the remainder of the area is Chernozemic. Table 8 contains a brief description of the predominant soil series, all of which are Orthic Brown Chernozems. More detailed information on the soils may be found in reports by Bowser and Kjearsgaard (1952) and Bowser et al. (1963).

The irrigation of these areas has moved considerable salt from the lower portions of the soil profiles to greater depths and corresponding SAR values have decreased (Committee of the Canada Department of Agriculture, 1960). However, hydraulic conductivities of soils in irrigated regions are lower than their dryland counterparts.

Distribution Systems and Farm Practices in the Study Areas

The Hays project is quite unique inasmuch as the entire

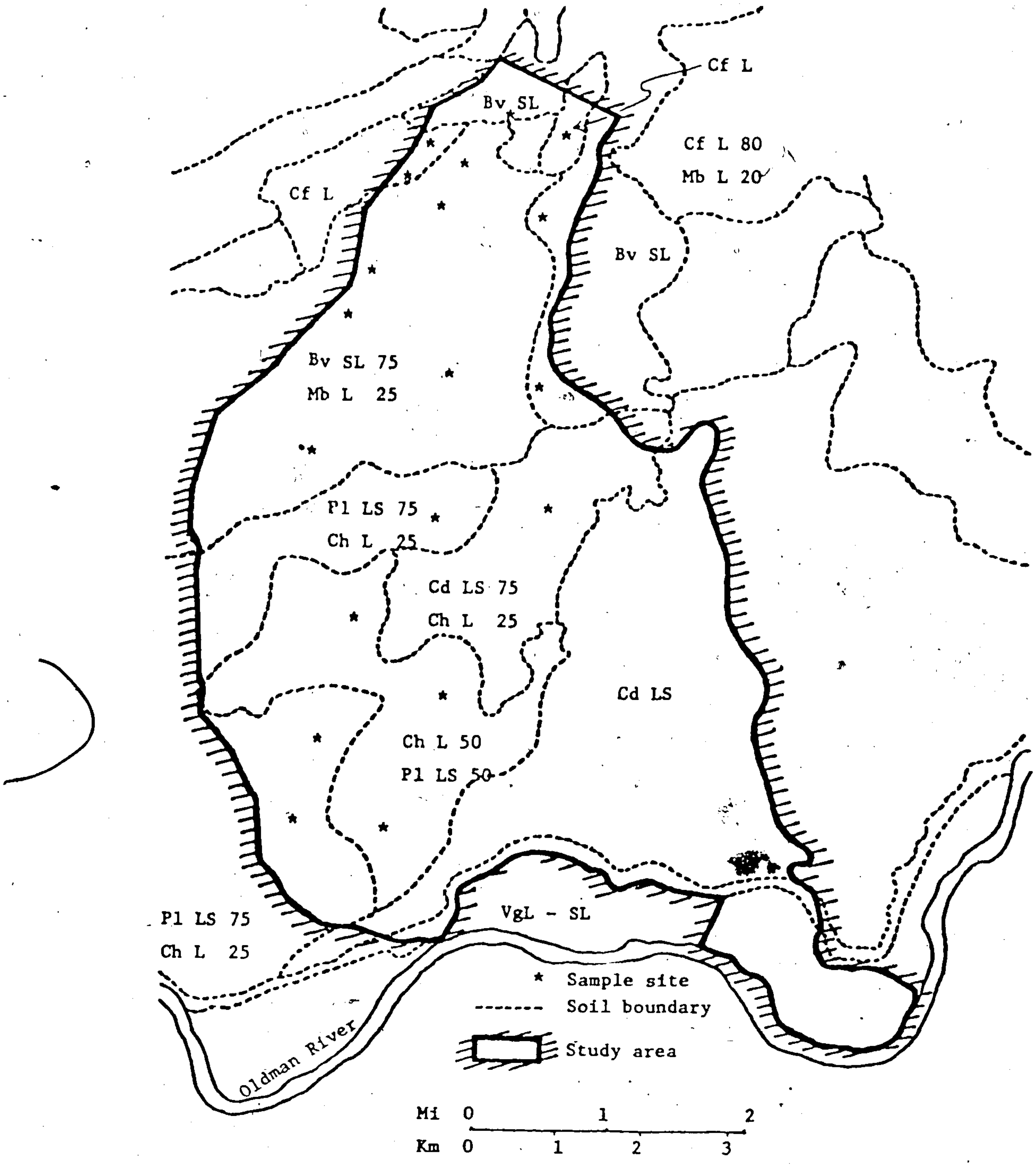


Figure 4. Soils of the Hays study area.

(See Table 8 for key to soils.)

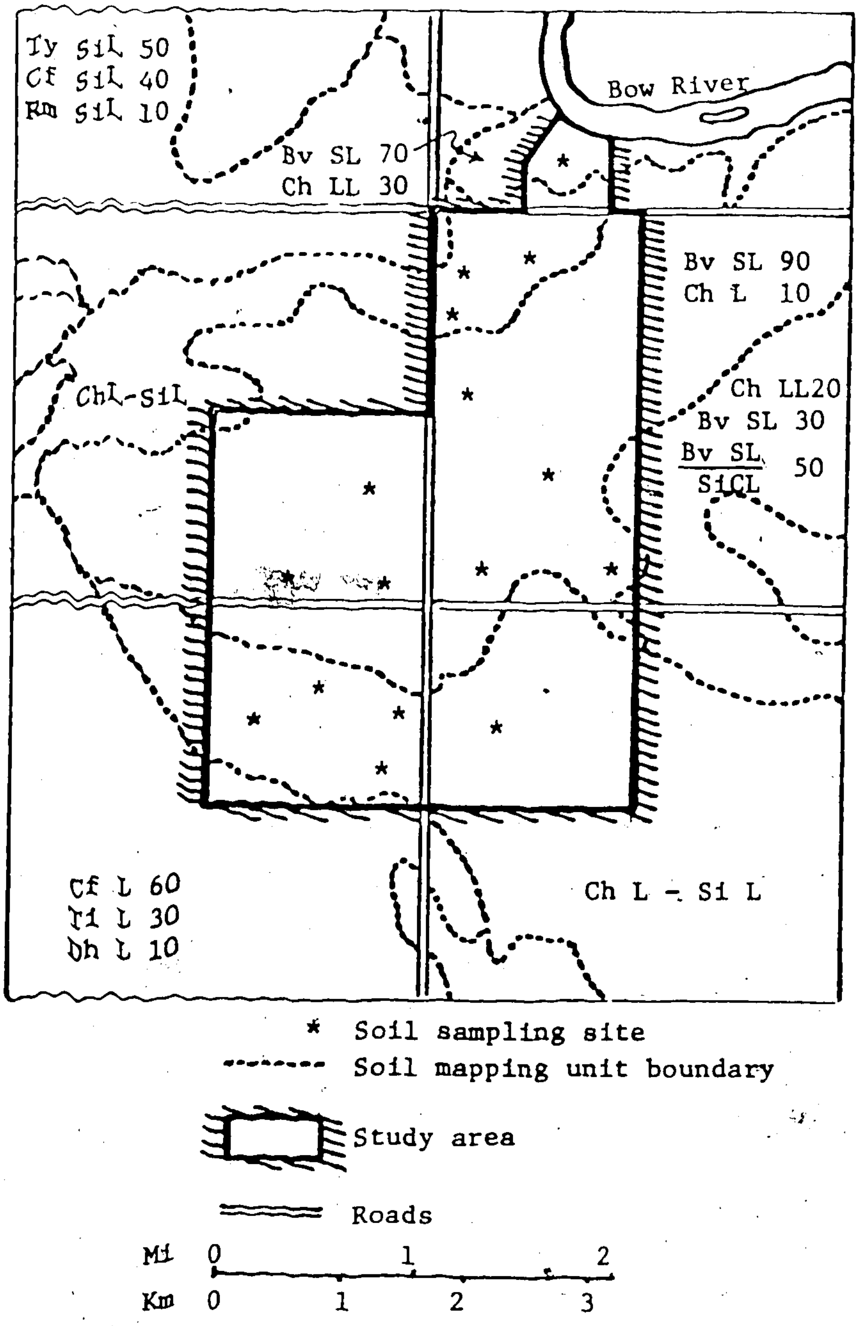


Figure 5. Soils of the Vauxhall study area.

Table 8. Predominant soil series in the study area.

Series	Type	Material	Typical Horizons*	Horizon Depth*	Former Name
Bingville (Bv)	Sandy loam	Alluvial aeolian	Ah	0-10 cm (0- 4 in)	Cavendish sandy loam
			AB	10-18 cm (4- 7 in)	
			Bm	18-50 cm (7-20 in)	
			BC	50-100 cm (20- 40 in)	
			Cca	at 100 cm (40 in)	
			Ck	at 125 cm (50 in)	
Cavendish (Cd)	Loamy sand	Alluvial aeolian	(Very similar to Bingville)		---
Chin (Ch)	Loam, silt loam	Lacustrine	Ah	0-10 cm (0- 4 in)	---
			Btj	10-20 cm (4- 8 in)	
			B	20-38 cm (8-15 in)	
			Cca	38-66 cm (15-26 in)	
			Csk	at 120 cm (48 in)	
Maleb (Mb)	Loam	Glacial till	Ah	0-10 cm (0- 4 in)	---
			Bt	10-30 cm (4-12 in)	
			Cca	30-45 cm (12-18 in)	
			Csk	45-60 cm (18-24 in)	
			C	at 90 cm (36 in)	
Purple Springs (Pl)	Loamy sand	Aeolian overlay	(Similar to Bingville but contact with till is at about 90 cm)		Shallow Cavendish loamy sand

* Information from Bowser et al. (1963).

irrigation system is designed specifically for flood-type irrigation. The water distribution and drainage network, even the roads, in this area follow contour lines. There is a built-in recycling mechanism for surface runoff waters which spill into downhill laterals and can be utilized again. Fields are long with a small artificially-leveled or natural gradient. Point rows are a common characteristic. As a result, all of the area until recently has been flood-irrigated; and as of 1976, only three farmers were using sprinklers. Crops are typically alfalfa, barley, and soft wheat with row crop acreage increasing proportionately to sprinkler irrigation.

In the growing season, sprinkler systems traverse the Vauxhall fields. Less than 120 ha (300 ac) are flood irrigated. The potato is "king" here in the "Potato Capital of the West" but sugar beets for the Taber sugar factory and alfalfa for the dehydration plant at Vauxhall occupy substantial acreage. Some barley and soft wheat are also grown.

Sites and Instrumentation

Irrigation and Drainage Water

Water quality data by itself is often insufficient for useful interpretation unless corresponding water quantity data are also provided. For this reason water level recorders were maintained at key locations to complement the chemical data obtained from water sample analyses. The names, locations, and instrumentations are given for each site in Table 9. Their relation to the distribution and drainage network can be seen in Figures 6 and 7.

Table 9. Water monitoring sites.

Name	Location	Instrumentation
Hays		
Lateral R In	NE 1/4-24-12-14	Water sample
Cudrak Supply	NE 1/4-14-12-14	Water sample
Cudrak Drain (Drain 231, upper)	SW 1/4-13-12-14	Recorder and water sample
Drain 231 (lower)	NW 1/4- 2-12-14	Recorder and water sample
Drain C	NW 1/4-21-11-13	Recorder and water sample
Lateral R Out	NE 1/4-17-11-13	Recorder and water sample
Drain T-A	NW 1/4-16-11-13	Recorder and water sample
Vauxhall		
Lateral C6	SW 1/4-10-14-16	Water sample
Pump Bypass	SW 1/4-11-14-16	Recorder
Lateral C6CA (Before Lagoons)	NW 1/4-11-14-16	Recorder and water sample
Lateral C6CA (After Lagoons)	NW 1/4-11-14-16	Water sample
Lateral E4	SE 1/4-14-14-16	Recorder
Lateral C6E	NW 1/4-14-14-16	Recorder
Lateral C6G1	NW 1/4-14-14-16	Recorder
Drain E Jnct.	NW 1/4-23-14-16	Water sample
Lateral C6C4	SE 1/4-26-14-16	Recorder and water sample
Drain E	SW 1/4-26-14-16	Recorder and water sample

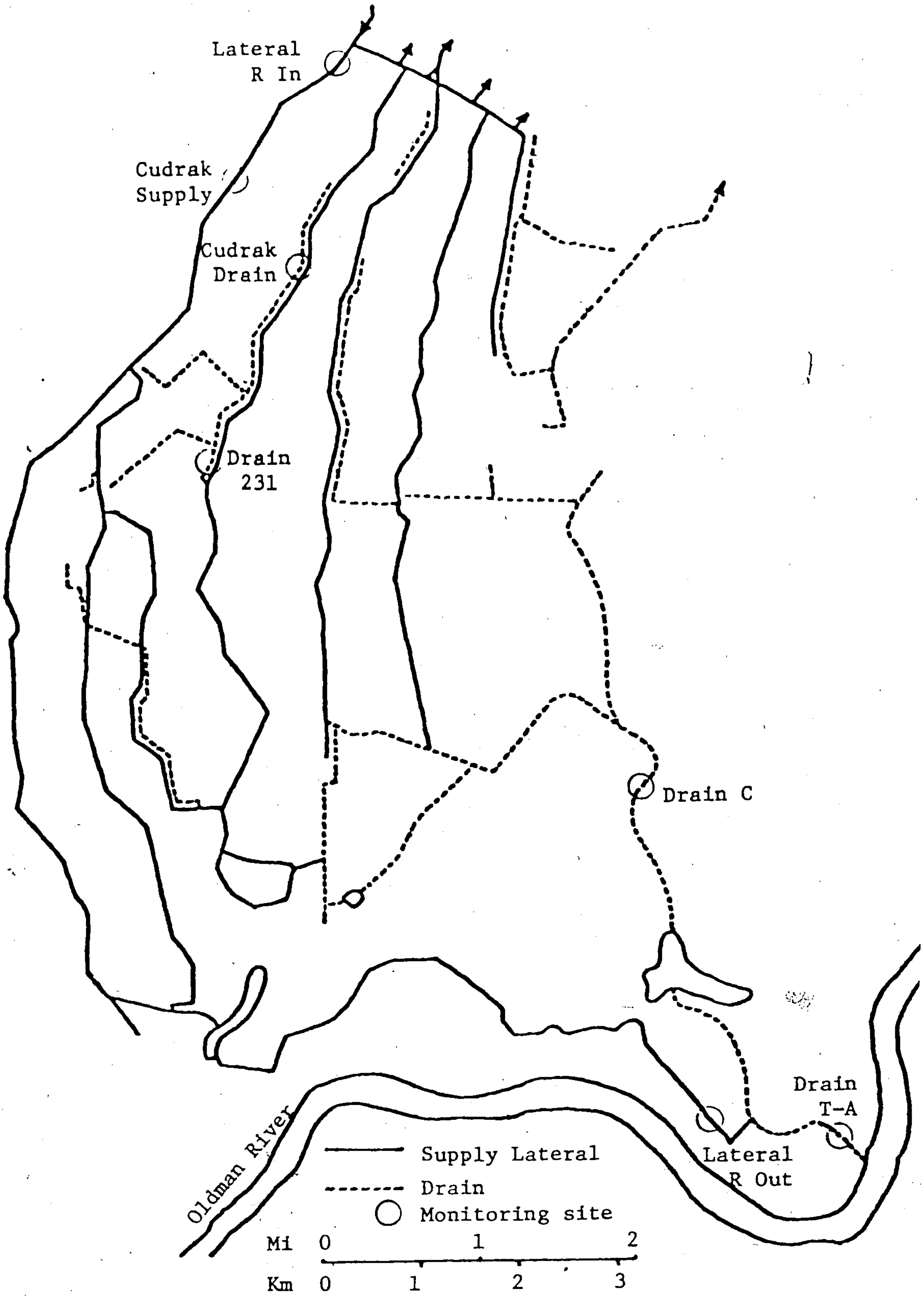


Figure 6. Water monitoring sites with respect to the distribution and drainage network in the Hays study area.

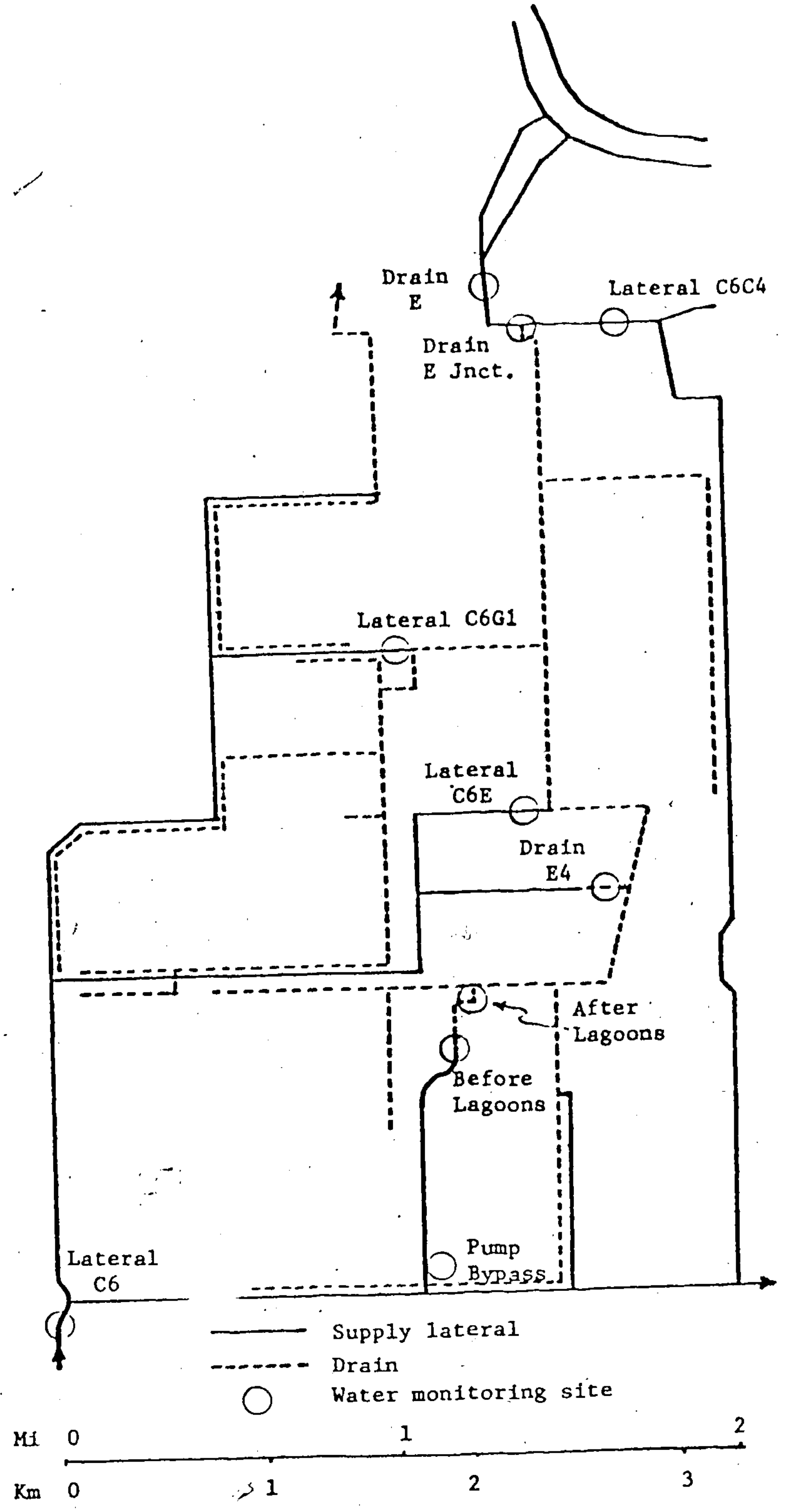


Figure 7. Water monitoring sites with respect to the distribution and drainage network in the Vauxhall area.

Each site had a definite purpose, as follows:

A. Hays

1. Lateral R In -- Water samples were taken here to determine the quality of the supply water as it entered the area. Quality characteristics downstream which differ from the supply may be attributed to point or diffuse water uses. The supply water, therefore, serves as a standard for comparison purposes. Because the supply of recorders was limited the flow was not monitored at this site. A large portion of this water does not enter the study area but is diverted northward. Because of the distribution system design, four recorders would be required to monitor the flow from Lateral R to the study area. Flow capacity of this lateral is approximately 300 ha-cm/hour (almost 300 cfs).
2. Cudrak Supply and Cudrak Drain -- The upper portion of Drain 231 collects the surface runoff from parts of Mr. F. Cudrak's farm. By comparing water samples of surface runoff and those taken at the point of farm supply, water quality changes attributable to flood-irrigation may be assessed at the individual farm level.
3. Drain 231 -- This drain services 9 fields covering some 500 ha (about 2 sections). Data here represent surface runoff, in terms of both quality and quantity, from a relatively large area. Because the laterals rather than drains collect the runoff in nearly every other portion of the Hays area, the proportion of delivery water that will appear as surface runoff is difficult to assess. Data from this drain are useful in estimating water application efficiency for the Hays area.

4. Drain C -- Past this point runs all the bypass water (spill water) and surface runoff water that was originally diverted from Lateral R for irrigation purposes. Drain C spills into Drain T-A, but part of the Drain C network is a small, shallow reservoir with a controlled discharge. In order to assess the water quality and quantity discharged from the irrigated area via Drain T-A, without the influence of the reservoir, sampling was necessary at this site.

5. Lateral R Out -- Flow passing this site is bypass water which has not been diverted for irrigation purposes. Quality fluctuations relative to the supply water are therefore solely the result of transport; the lateral receives no surface runoff [illegible] because it is located on topographic highs, probably receives little or no seepage water.

6. Drain T-A -- This is the point of discharge of surface-transported irrigation return flow to the Oldman River. When data from this site are compared with the supply water, quality fluctuations attributable to irrigation practices (mainly flood-irrigation) can be determined; and the impact of the return flow on downstream use can be predicted.

B. Vauxhall

1. Lateral C6 -- As in the Hays study, the quantity of supply water to the Vauxhall area is difficult to monitor. Quality was regularly monitored here to serve as a standard when assessing quality fluctuations in the rest of the area.

2. Pump Bypass -- Maintaining flow to a pump in excess of the rate of pumping is necessary to avoid the loss of pump-prime. When air enters the intake pipe, pumping will cease. The excess water, supplied

over-and-above pump needs is sometimes spilled into a surface drain. This site monitors the quantity of such a flow.

3. Before Lagoons (Lateral C6CA) -- Similarly to the pump bypass described above, water is spilled at this site when supplied in excess of pump needs. The water in the surface drain then runs past 2 hog-sewage lagoons situated some 20 feet to the west. The amount of irrigation water spilled is of interest as is the quality of the water before the lagoons. Both were monitored.
4. After Lagoons (Lateral C6CA) -- Approximately 500 m (1/3 mi) downstream from the Before Lagoons site, water quality samples were taken to determine whether any effect of the sewage ponds on the passing spill water was noticeable; in particular to what degree the nutrient status of the drainage water was enhanced by this potential point source.
5. Lateral E4 -- This site also monitored pump bypass water. Though the ditch was large, the flow was small and the site was abandoned in 1976.
6. Lateral C6E -- Bypass water entered the surface drainage network at the tail end of this supply lateral. A recorder was used in monitoring the flow that spilled into the drain.
7. Lateral C6G1 -- The situation at this site is comparable to that of Lateral C6E.
8. Lateral C6C4 -- A portion of the supply water, which was diverted from Lateral C6 to service the farms along the eastern border of the study area, exceeds farm demands. The unutilized flow runs past this point to spill into Drain E. Very little to no surface runoff water or drainage water is transported with the bypass water. Comparable

to Lateral R Out, Hays, water quality at this site is expected to differ only slightly from that of the supply. Differences would be attributable to processes associated with transport -- not drainage.

9. Drain E Jnct. (the junction of Drain E and Lateral C6C4) collective surface runoff, subsurface flow to open drains, and spill water from previously mentioned Vauxhall sites passes here. Quality was monitored directly; quantity was determined by difference between Drain E and Lateral C6C4 flows.

10. Drain E -- Water passing this site is the collective surface return flow from the Vauxhall study area. The Bow River is the receiving water body. When the quality of the Drain E flow is compared with the supply water, quality fluctuations attributable to irrigation practices (mainly sprinkler-irrigation) can be assessed. The impact of the return flow on downstream use can be estimated.

Precipitation

Rain gauges were installed at the north and south ends of both study areas. Each area had one Canadian standard rain gauge and one improvised rain gauge of near standard diame[illegible] Precipitation was recorded weekly.

Fertilizer Application

Rates of fertilizer application were determined by interviewing each farmer. Actual N and actual P values were calculated from reported weights and analyses.

Crops

Estimates of the areal extent of the various crops were obtained by on-farm interviews.

Sampling and Sample Preparation

Water

Except for the examination of surface runoff from the Cudrak farm, all water samples were taken biweekly. Half-liter polyethylene bottles were filled by the single vertical dip method. The runoff water from the Cudrak farm was sampled each year of the study over a substantial portion of a day at hourly intervals. The corresponding supply water was also sampled but less frequently. In 1976, several samples of this runoff water were taken farther [illegible] udrak Drain (which is actually the upper reach of Drain 231) to [illegible] the effect of the downstream aquatic growth and slow transport found in Drain 231 on water quality. These samples were taken just north of the regular Drain 231 site so as to avoid mixing with waters coming from other fields to the southwest.

At each site, water samples were taken in duplicate. One of the samples was immediately filtered through a 0.45 um filter. Two drops of toluene were added to each bottle to inhibit biological activity. The samples were [illegible] placed in a container for transport. In 1976, an ice-cooled chest was used, whereas in 1975 no artificial cooling was supplied. The samples were transported to the laboratory and kept at a temperature of approximately 4°C. Nitrate-nitrogen and ortho-phosphorus were determined during the first and second day of storage, respectively. Other analyses were run as soon as possible.

Soil

Soil samples were taken in the spring of 1975 before fields were planted, and at corresponding locations in the fall of 1976 after fields were harvested. Samples were taken with the aid of a coring truck; most were cores, but many of the holes during the 1975 sampling had to be augered The soil sampling locations are indicated on Figures [illegible] and 5. Because harvesting was not yet completed in all areas, two of the samples in the Vauxhall area could not be repeated in 1976.

Samples for chemical determinations were taken at each site by increments of depth: 0 to 30.5 cm (0 to 1 ft), 30.5 to 61 cm (1 to 2 ft), 61 to 91.5 cm (2 to 3 ft), and 91.5 to 183 cm (3 to 6 ft). One sample from 183 to 244 cm (6 to 8 ft) was taken per soil mapping unit. Samples for physical analyses were taken by horizons in the fall of 1976. In 1975 the samples were oven dried at 105°C. As a result the ammonium and available phosphorus data presented for 1975 are expected to be high. In 1976 the soil samples were air dried. Barker (1974) in a paper on nitrate determination states that "oven-drying may be acceptable for soils. . ." but that air-drying is preferred. Some of the nitrate associated with organic compounds in the soil would be lost on oven-drying, but Orthic Brown Chernozems have low organic content. (See Appendix I, for results of these analyses.)

Analytical Methods

Chemical

Table 10 gives the analytical methods used in this study for the examination of the water and soil chemistry. Further comment is needed to clarify some points. Because of extremely low nitrate contents

Table 10. Analytical procedures for chemical constituents of soil and water*.

Analysis	Reference	Sample
Ammonium	Bremner (1965)	Soil extract
Bicarbonate and carbonate	Bower and Wilcox (1965)	Water, and soil saturation extract
Calcium and magnesium	Chang and van Schaik (1965)	Water, and soil saturation extract
Cation exchange capacity	Chapman (1965)	Soil samples
Chloride	Technicon Industrial Systems (1974)	Water, and soil saturation extract
Electrical conductivity	Bower and Wilcox (1965)	Water, and soil saturation extract
Exchangeable sodium	United States Salinity Laboratory Staff (1954)	Soil samples
Magnesium	Steckel and Flannery (1965)	Water, and soil saturation extract
Nitrate	Harper (1924)	Soil extract
Nitrate	Sommerfeldt <u>et al.</u> (1971)	Water
Nitrogen (total)	Lepper (1950)	Soil Samples
pH	United States Salinity Laboratory Staff (1954)	Water, and saturated soil paste
Potassium	Technicon Autoanalyzer Methodology (1963)	Water, and soil saturation extract

Table 10. (Continued)

Analysis	Reference	Sample
Phosphorus (available)	Olsen _et al._ (1954)	Soil extract
Phosphorus (ortho-PO_4, acid-hydrolyzable-PO_4, and total-PO_4 by persulfate)	American Water Works Association (1971)	Water
Phosphorus (total-PO_4 by perchloric)	American Public Health Association (1971)	Water
Sodium	Chang and van Schaik (1965)	Water, and soil saturation extract
Sulfate	Technicon Industrial Systems (1972)	Water, and soil saturation extract

* Information, excluding total-PO_4 by perchloric, from Carefoot (1976).

in the surface waters encountered, the nitrate electrode was operated to determine concentrations just below recommended limits. The author feels that the relative nitrate concentrations are well represented by the derived data and that quantitatively, the data are quite accurate. The perchloric acid method for total-P determination in water samples was employed in 1976 because of suggestions that the persulfate digestion method was less efficient in the recovery of particulate-P (O'Connor and Syers, 1975). To maintain continuity the persulfate method was also continued, and a relationship of total-P by persulfate to total-P by perchloric acid was obtained by regression analysis (Appendix II).

A large portion of the chemical analyses were performed by the Soil Science Service Laboratory Staff.

Physical

A. Water

Rating curves were utilized to extrapolate discharges from continuous water level data. At each site where water level recorders were installed, flows were measured at various times, more importantly at different water heights above a datum. These measured heights were plotted against flows to provide a standard height-discharge curve -- a rating curve. Mean daily heights were determined for each site by means of a planimeter, and flows were extrapolated from the rating curve. Water Survey of Canada provided the flow records for Drain T-A and Drain E.

Suspended load was determined by filtration. 400 to 500 ml were drawn through a previously weighed, 0.45 um filter. Each filter was placed in a separate carrying container. The increased dry weight of the filter paper was attributed to the suspended load. The water volume of each filtered sample was measured at the laboratory. Suspended load was determined in 1976 only.

Water temperatures were taken to the nearest 0.5°C at the time of sampling.

B. Soil

1/3-bar and 15-bar percentages were determined as described by the United States Salinity Laboratory Staff (1954). Dry bulk density of the soils was determined by oven drying and weighing a known volume of the core samples. A peroxide pretreatment and the use of pure Na-hexametaphosphate instead of calgon were variations from Day's method (1965), in performing the particle size analysis for this study. Following the theory presented by Klute (1965), disturbed hydraulic conductivities were determined on soil ground to pass a 2 mm sieve. Soil columns were mechanically tamped 200 times, then saturated under vacuum before being mounted on a constant head apparatus.

Statistical Analyses

To determine whether there were any statistically significant differences between total dissolved salt concentrations of the supply water and those of the corresponding return flow, t-tests were applied to compare mean values. Student's t-tests were also applied in assessing whether there were any real differences in salt content between the 1975 and 1976 soil samplings at matching depths. Correlation coefficients, for the linear correlation of flow with the studied components of water quality, were calculated to aid in the assessment of causal relationships.

Statistical procedures as outlined by Zalik (1975) were used in the above determinations.

RESULTS AND DISCUSSION

Water

Potential Pollution from Eutrophication

A. Phosphorus

(a) Hays area

Tables 11 and 12 present data on phosphorus discharges from the Hays area. Tables 13 and 14, respectively, give results for the 1975 and 1976 studies at the Cudrak farm. For convenience, nitrate values are also contained in the tables. The total-P concentration in the runoff from the Cudrak farm in the 1975 study was 3 times that of the supply water; in 1976 it was 13 times. Though the data are scant, it would appear that the concentration of the supply water has little effect on the concentration in the runoff water. Rather, an equilibrium of soil and sediment phosphorus with the applied irrigation water controls the total-P concentration in surface runoff at this site.

In light of the fact that runoff from the study farm had total-P concentrations near 0.180 ppm in both 1975 and 1976, it is surprising that the corresponding total-P concentrations in the return flow from the entire Hays study area were only 0.069 and 0.061 ppm. Why are the total-P concentrations greater on an individual farm basis than those from the entire area? These low values noted in the discharge to the Oldman River are a result of two factors: (1) the settling of phosphorus-bearing sediments, and (2) the dilution of phosphorus in

Table 11. Phosphorus and nitrate-nitrogen concentrations in irrigation and drainage waters in the Hays area.

	Lateral R In (supply water)		Drain T-A (return flow)		Lateral R Out (a bypass water)		Drain 231 (a surface runoff)		Drain C (bypass and runoff)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1975										
Total-P* (ppm)	0.052	0.021	0.069	0.017	0.066	0.020	0.153	0.077	0.067	0.026
Ortho-P (ppm)	0.019	0.011	0.022	0.010	0.025	0.007	0.069	0.062	0.024	0.014
NO_3-N (ppm)	0.131	0.073	0.115	0.054	0.090	0.060	0.130	0.155	0.090	0.049
1976										
Total-P* (ppm)	0.030	0.009	0.061	0.035	0.031	0.010	0.058	0.038	0.057	0.014
Ortho-P (ppm)	0.007	0.005	0.010	0.004	0.006	0.004	0.026	0.025	0.021	0.024
NO_3-N (ppm)	0.173	[illegible]	0.194	0.108	0.177	0.108	0.185	0.125	0.167	0.089

* Total-P by the persulfate method.

Table 12. Total seasonal phosphorus and nitrate-nitrogen loads in irrigation and drainage waters in the Hays area.

	Lateral R In (supply water)	Drain T-A (return flow)	Lateral R Out (a bypass water)	Drain 231 (a surface runoff)	Drain C (bypass and runoff)
1975					
Total-P* (kg)	1,350	510	90	50	480
Ortho-P (kg)	360	160	30	20	190
NO_3-N (kg)	3,150	910	120	80	710
1976					
Total-P* (kg)	910	490	80	20	480
Ortho-P (kg)	150	100	10	10	160
NO_3-N (kg)	4,440	1,510	360	90	1,220

* Total-P by persulfate method.

Table 13. Phosphorus and nitrate-nitrogen concentrations in the supply water, surface runoff, and downstream drainage water associated with the Cudrak farm study, 1975.

Sample Description	Total-P* (ppm)		Ortho-P (ppm)		NO_3-N (ppm)	
	Mean	SD	Mean	SD	Mean	SD
Supply water**	0.058	--	0.028	--	0.110	--
Surface runoff (crop: alfalfa)	0.177	0.085	0.111	0.026	1.380	0.763
Drain 231 (1975 mean)	0.153	0.071	0.069	0.062	0.130	0.155

* Total-P by the persulfate method.

** Only one sample taken.

Table 14. Phosphorus and nitrate-nitrogen concentrations in the supply water, surface runoff, and downstream drainage water associated with the Cudrak farm study, 1976.

Sample Description	Total-P by Persulfate (ppm)		Total-P by Perchloric (ppm)		Ortho-P (ppm)		NO_3-N (ppm)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Supply water	0.014	—*	0.020	—	0.000	—	0.093	—
Surface runoff (crop: wheat)	0.185	0.044	0.301	0.061	0.081	0.017	1.601	0.327
Runoff downstream in Drain 231	0.067	—	0.086	—	0.022	—	0.058	—
Drain 231 (1976 mean)	0.058	0.038	—**	—	0.026	0.025	0.185	0.125

* Only two samples taken.

** Not all samples were analyzed using perchloric digestion.

runoff waters by purer bypass water. Downstream samples of the runoff from the Cudrak farm (taken just north of the usual Drain 231 site) bore results that showed a significant decrease in both total-P and suspended load with respect to the upstream flows. Since flow velocities in Drain 231 are in the order of 25 to 50 cm/sec and depths of flow equal 15 to 20 cm roughly, there is ample time for the settling of suspended materials along the 2.5 km stretch of drain. There is little turbulence to maintain the suspension of soil particles; suspended load and hence total-P therefore decrease. Runoff water from the study farm eventually spills into a supply lateral as does all runoff in the Hays area, and chemical constituents are diluted by the relatively pure receiving stream. As a result, total-P concentrations are decreased though the phosphorus load is not decreased. Thus, P concentrations from point sources may be high while at the same time gross exports may be unexpectedly low.

Less than 0.2 kg of total-P/cultivated ha/yr was discharged into the Oldman River from the Hays area in the Drain T-A return flow. This unexpectedly low value represents 1.1 percent of the gross yearly phosphorus inputs to the area (gross inputs equal total-P in supply water plus fertilizer P applications, Tables 12 and 15). The phosphorus concentration in the return flow was found to be greater than that in the supply water, 1.3 fold in 1975 and 2.0 fold in 1976, but because the quantities of return water were less than supply quantities, total phosphorus exports from the Hays area amounted to less than the total phosphorus imported in the supply water. Of the 500 or so kilograms of total-P exported from the Hays area, 23 percent in 1975 and 50 percent in 1976, had a different origin than that of the supply water. These

Table 15. Fertilizer applications in the Hays and Vauxhall areas, 1975.

Location	Total Fertilizer Applied (kg/ha)	Actual N Applied (kg/ha)	Actual P Applied (kg/ha)	Cultivated Area (ha)	Total Fertilizer Applied (kg)
Hays watershed	132	37	17	2,560	3.39×10^5
Vauxhall watershed	237	45	35	1,090	2.58×10^5

figures represent maximum contributions possible from any one source such as fertilizers.

Some researchers estimate fertilizer P contributions to drainage waters by the relative increase in ortho-P. The 1975 and 1976 increases in the irrigation return flow over-and-above supply concentrations averaged 0.003 ppm both years -- a small fraction of the total-P increases. On this basis, little phosphorus of fertilizer origin is discharged into the Oldman River from this area.

(b) Vauxhall area

Tables 16 and 17 present phosphorus discharges from the Vauxhall area. The tabulations show some surprising results in the study of nutrient levels of spill water at the Before and After Lagoons sites in the Vauxhall area. In 1975 total-P concentrations at the monitoring point downstream from the lagoons ranged from 1.24 to 16.3 times those of the upstream site. The wide variation may be attributed in the main to differences in flow rates at the time of sampling. However, the data indicate a definite increase in total phosphorus. The average increase in total-P concentration of 2.0 fold, as determined in 1976, was obtained without any sampling at low flows.

Because the polyethylene lined lagoons are some 7 meters from the drainage channel, and because the sewage is not discharged into the drain but rather is used for irrigation of an adjacent field, an increase in phosphorus in the passing water was not expected. Calculated values showed an increased total-P load of approximately 60 kg in 1975 and 30 kg in 1976 for the respective irrigation seasons. Since hog feed is supplemented with phosphorus to maintain the proper calcium to phosphorus

Table 16. Phosphorus and nitrate-nitrogen concentrations in irrigation and drainage waters in the Vauxhall area.

	Lateral C6 (supply water)		Drain E (return flow)		Lateral C6C4 (a bypass water)		Drain E Jnct. (bypass and drainage)		Before Lagoons		After Lagoons	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1975												
Total-P* (ppm)	0.068	0.040	0.070	0.029	0.055	0.009	0.088	0.030	0.057	0.014	0.412	0.444
Ortho-P (ppm)	0.018	0.008	0.018	0.010	0.022	0.010	0.024	0.011	0.012	0.004	0.028	0.000
NO_3-N (ppm)	0.063	0.019	0.133	0.063	0.138	0.056	0.108	0.050	0.156	0.104	0.668	0.751
1976												
Total-P* (ppm)	0.047	0.027	0.076	0.059	0.047	0.059	0.079		0.043	0.016	0.072	0.054
Ortho-P (ppm)	0.007	0.005	0.027	0.030	0.018	0.027	0.035		.008	0.004	0.025	0.017
NO_3-N (ppm)	0.184	0.099	0.277	0.346	0.290	0.194	0.216		0.149	0.076	0.193	0.112

* Total-P by the persulfate method.

Table 17. Total seasonal phosphorus and nitrate-nitrogen loads in irrigation and drainage waters in the Vauxhall area.

	Lateral C6 (supply water)	Drain E (return flow)	Lateral C6C4 (a bypass water)	Drain E Junct. (bypass and drainage)	Before Lagoons	After Lagoons
<u>1975</u>						
Total-P* (kg)	1,040	260	110	130	9	69
Ortho-P (kg)	240	40	50	40	2	4
NO_3-N (kg)	830	450	250	130	22	111
<u>1976</u>						
Total-P* (kg)	820	350	80	210	27	55
Ortho-P (kg)	90	120	30	80	5	16
NO_3-N (kg)	2,970	1,120	330	520	90	114

* Total-P by the persulfate method.

ratio required for bone development, the sewage effluent was expected to be high in phosphorus. A small leak could cause the discovered results though no surface flow was apparent. To add to the enigma, the suspended load was consistently lower at the After Lagoons site.

Though the lagoons made noticeable contributions to the P budget of the Vauxhall study area, the total phosphorus exports from the entire tract remained low as was expected: 1975 and 1976 exports were less than 0.3 kg/ha/yr. Surface runoff of sprinkler-applied water was minimal, in fact never was noticed by the author during the two-year study. Consequently, only a minute amount of eroded phosphorus-rich sediment in drainage channels was anticipated. This assumption, however, did not account for wind erosion which partially filled the eastern channels of the Vauxhall tract with topsoil during the early parts of 1976. Thus, the 1976 total-P exports of 350 kg were found to be 1.4 times those of the corresponding 1975 period. These exports amount to <0.9 percent of the gross inputs into the area from irrigation and fertilization. Imports in the supply water exceeded exports in the flow past the Drain E monitoring site. As was the case in the Hays study basin, only a fraction of the P discharge may be attributable to fertilizer usage in 1975. Ortho-P accounted for 20 of the 29 ppb total-P increase in 1976. Again, this increase is mainly attributable to phosphorus rich surface soils which were wind-deposited in the distribution network. There was no increase of ortho-P in the return flow concentration over the supply water in 1975.

(c) Algal biomass equivalence

What would be the potential production of algal biomass per cubic meter of river flow as a result of the return flows from the two areas? To help answer that question, a new term (and calculation) is coined here -- the "algal biomass equivalence." This calculation gives an estimate of the maximum primary production arising from the discharged P. Using an average percent P (0.42 percent on an oven-dry basis) for the phosphorus content of algae equal to the mean percentage of 15 species as given by the National Technical Advisory Committee (1968), and assuming that all the total-P is converted to algal biomass, then one can calculate the maximum biomass that could be produced from a certain quantity of nutrient P. The algal biomass equivalence of the P discharged from the Hays area is approximately 120 t of algae, while that from the Vauxhall area would be approximately 70 t on a dry weight basis.

From the mean flow of the receiving waters, this algal biomass equivalence can be converted to a maximum increase in algae per cubic meter of receiving water. Using the 1971-1975 mean flows from May through September for the Oldman and Bow Rivers (Water Survey of Canada, 1971-1975 inclusive), the increase in algae in the receiving rivers would be 0.05 g/m^3 and 0.04 g/m^3 for the Oldman and Bow, respectively, as a result of the phosphorus in the studied irrigation return flows. Using data from Lake Sebasticook, Maine as cited by Mackenthun (1969), a total algal weight of 12.5 g/m^3 can be calculated as follows for the lake's polluted waters. First, let us assume a bulk density of algae equal to that of water. The water contained on the average 600 cells/ml and 15 ppm on a volume basis. Mackenthun used 500 cells/ml as a division for polluted and non-polluted waters. Since on a dry weight basis algae weigh approximately 1/10 their wet weight, a cubic meter of water with a cell count of 500/ml would contain 1.04 g of algae on a dry weight basis, or 10.4 g/m^3 on a wet weight basis, and the 600 cells/ml would thus correspond to 12.5 g/m^3.

The algal biomass equivalence values indicate that only small increases over-and-above natural levels in algal growth could result from the irrigation return flows. The average concentration of total-P discharged from the study areas are lower than those required to avoid nuisance algal growths (Tables 7, 11, and 16). Thus, the phosphorus discharged in the irrigation return flow from the Hays and Vauxhall study areas are minimal and should not be considered as adversely eutrophic.

B. Nitrate nitrogen

(a) Hays area

The data on N are presented in Tables 11 and 12. As in the case of phosphates, but more dramatically, the nitrate load of the surface runoff from the Cudrak farm decreased as the water flowed 2.5 km down Drain 231. Though NO_3-N concentrations in the surface runoff from the study farm averaged 1.60 ppm in 1976, corresponding concentrations of the water downstream averaged 0.26 ppm -- close to the yearly average of 0.21 ppm for Drain 231. Similar results were found in 1975 for the farm runoff and the yearly average nitrate concentration for Drain 231. Because of the large interface between the channel and the flowing water, transformations of the NO_3-N to a reduced form possibly occurred in the sediments, as the literature would indicate is possible. Further study would be necessary to explain the nitrate losses.

Again, the point source supply of a nutrient element may be high while from the overall area, concentrations are low. In 1975 the NO_3-N concentration in Drain T-A waters averaged 0.115 ppm and in 1976, 0.194 ppm. Nitrate exports amounted to 0.4 kg of N/ha in 1975 and 0.6 kg N/ha in 1976. These values are equivalent to <1.0 percent of the 1975 gross inputs and <1.6 percent of the 1976 gross inputs.

(b) Vauxhall area

Data for N in the Vauxhall area are given in Tables 16 and 17. The lagoons that were mentioned in the phosphorus section also acted as a source of NO_3-N. This was expected, and was the primary reason for studying the water quality at the Before and After Lagoons sites. Concentrations of NO_3-N at the After Lagoons site were 4.3 times those of the Before Lagoons site in 1975 and 1.3 times greater in 1976. Again, flow volumes made the primary difference in noted concentrations.

The calculated values of total NO_3-N exports from the entire Vauxhall study area amounted to <1.0 percent of total inputs in 1975 and <2.2 percent in 1976. In both years, imports in the supply water were in excess of exports in the surface return flows.

(c) Algal biomass equivalence

Using the procedure described in the previous section on phosphorus, the potential algal biomass produced from a given amount of nitrate may be calculated. Average N content of algae equals 4.2 percent on a dry weight basis, as calculated from data given by the National Technical Advisory Committee (1968). For the Hays area, the 914 kg of

NO_3-N discharged into the Oldman River is thus equivalent to a potential 22 t of algae. The 1976 equivalence for the Hays area equals 36 t. For the average flow of the Oldman River (183 m^3/sec) from May through September, the 1975 NO_3-N discharge is equivalent to 0.009 g of algae/m^3; for 1976, 0.015 g/m^3. The algal biomass equivalence of the exported NO_3-N in the Drain E return flow equals 11 t for the 1975 irrigation season and 27 t in 1976. These amounts of algae equal 0.006 g/m^3 and 0.014 g/m^3 of flow in the Bow River. Nitrate discharged in the irrigation return flow from the Vauxhall and Hays study areas should not therefore be considered as adversely eutrophic.

Potential Pollution from Sediment

A. Hays area

The supply water for the Hays area comes from Reservoir #1 which has a capacity of some 12 million cubic meters (Research Council of Alberta, 1970). The water discharged from the reservoir into Lateral R is relatively free of suspended materials, averaging 8.5 mg/l in 1976 (Table 18). In the surface runoff from the Cudrak farm, the mean concentration of suspended materials was 86.4 mg/l in the 1976 study. A 10-fold increase in suspended load concentration is not surprising for a flood-irrigated area. However, the suspended load from the entire Hays study area was lower than from this point source. As depicted in Figure 8, the maximum suspended load in Drain T-A was 64 mg/l in late May; the average for the irrigation year was 33.5 mg/l. This concentration is significantly lower than that in the surface runoff from the Cudrak farm.

Table 18. Suspended load of irrigation and drainage waters in the Hays area, 1976.

Site and Description	Mean (mg/l)	Standard Deviation (mg/l)
Lateral R In (supply water)	8.5	3.7
Drain T-A (return flow)	34	17
Lateral R Out (a bypass water)	25	11
Drain 231 (a runoff water)	14	9.4
Drain C (bypass + runoff)	35	15
Cudrak Supply (a supply water)	11	--*
Cudrak Drain (a runoff water)	86	29
Cudrak Drain Downstream (a runoff water)	38	--

* Only two samples.

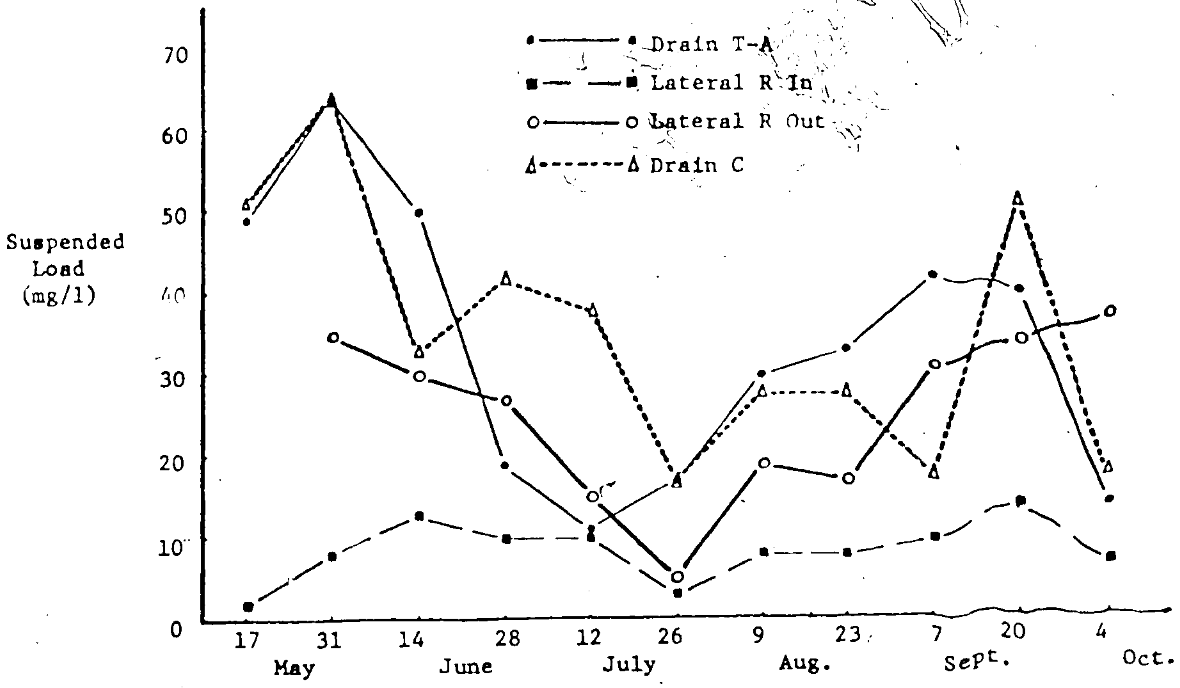


Figure 8. Sediment concentrations in the irrigation and drainage waters in the Hays study area, 1976.

Both the settling of particulate matter and the dilution of runoff with bypass water decrease the suspended load concentration. Some 2.5 km from the point of runoff, the suspended load of the surface drainage water from the Cudrak farm averaged 38 mg/l, indicating that over half of the particulate material had settled out of suspension. Spillage of [illegible] surface drainage water into the more pure bypass water would result in further dilution of the suspended load.

It was expected that the small pond connecting Drain C and Drain T-A (Figure 6) would cause a significant reduction in sediment loads in Drain T-A. This was generally not the case as noted by the relative sediment concentrations depicted in Figure 8. In fact there is less than 2 percent difference in the Drain C and Drain T-A mean values. Either the pond had a poor sediment trap efficiency or the suspended materials were fine enough to be maintained in suspension by Brownian motion. Both are definite possibilities.

Compared to the quality criteria in Table 7, the suspended load concentration from the Hays area offers a relatively high level of protection for the growth of desirable freshwater life. However, water treatment would be required before the return flow could be used by food-processing industries and municipalities.

Although subjective in nature, the following group of observations seem important. [illegible] but one of the 1976 sampling trips, the Oldman River appeared [illegible] than the Drain T-A flow as the river passed over a broad shoal near the mouth of the drain. Peak turbidity in the river occurred shortly after heavy rains the first week of August, 1976. Even the Waterton and Belly Rivers, which are tributaries of the Oldman, appeared brown at this period though they are situated far upstream from major return flow sites. Pertaining to recreation and wildlife, the fish that enjoy the small fresh water crustacea in the return flow attract a continual flow of local fishermen. The small sediment load seems to bother neither the fish nor the fishermen.

Thus, the sediment load discharged in the irrigation return flow does not seem detrimental to the use of the Oldman River. On the contrary, when considering sediment loads, the return water appears to enhance the quality of the river.

B. Vauxhall area

Considering the fact that the Vauxhall area is nearly all sprinkler-irrigated while the Hays area is almost all flood-irrigated, it was surprising to find that exports of suspended materials from the Vauxhall tract (195 kg/ha) were nearly 2 times those from the Hays area per cultivated hectare. This apparent contradiction does have a well defined basis, however. The supplying reservoir for the Vauxhall region is some 50 km from the area. Bank sloughing and erosion, and soil drifting, would be expected to add to the suspended load of the supply canal. This would explain why approximately 900 t (840 kg/ha) of sediment was imported into the Vauxhall area. There was a net accumulation of 700 t of sediment in the Vauxhall area in 1976, while in the Hays area there was a small net export -- the sprinkler-irrigated area "filtered out" a large portion of the sediment while the flood-irrigated area added to the suspended load. Comparing supply and return flow concentrations (Table 19), all of the sediment discharged in the return flow from the Vauxhall area could have originated in the supply water

Table 19. Suspended load of irrigation and drainage waters in the Vauxhall area.

Site and Description	Mean (mg/l)	Standard Deviation (mg/l)
Lateral C6 (supply water)	49	42
Drain E (return flow)	44	20
Lateral C6C4 (a bypass water)	48	88 *
Drain E Jnct. (bypass + seepage)	39	24
Before Lagoons	24	11
After Lagoons	15	12

* Excluding one exceptionally high value, the standard deviation would equal 12 mg/l.

whereas most of the sediment in the Hays return flow came from some other source -- presumably surface runoff.

Two other factors played a role in increasing the suspended load of the return flow from the Vauxhall study area. The maximum suspended load occurred in Lateral C6C4 (Figure 9), a channel which has low flow velocity and receives little to no drainage water. This surprising high concentration of 311 mg/l, measured on the first sampling trip in May, was a result of windblown topsoil accumulations in the lateral. The second factor was dredging of Drain E just before its junction with Lateral C6C4, which increased post-July 12 suspended loads as monitored at the Drain E junction site (Figure 9).

Comparing the Drain E suspended load concentration of 43.5 mg/l with the water quality criteria, the sediment content of the water falls between high and moderate levels of protection for aquatic life. Municipalities and food-processing industries require clearer water than that discharged in the Drain E return flow. Some water treatment prior to use would be required by these users.

Potential Pollution from Dissolved Salts

A. Total dissolved salts

(a) Hays area

The 1975 and 1976 data on total dissolved salts [illegible] the Hays area are contained in Tables [illegible] and 21. Specific salt data are also included for convenience. With respect to the total dissolved salts, a t-test showed no significant difference between the supply water and the irrigation return flow in 1975. However, in 1976 there was a significant

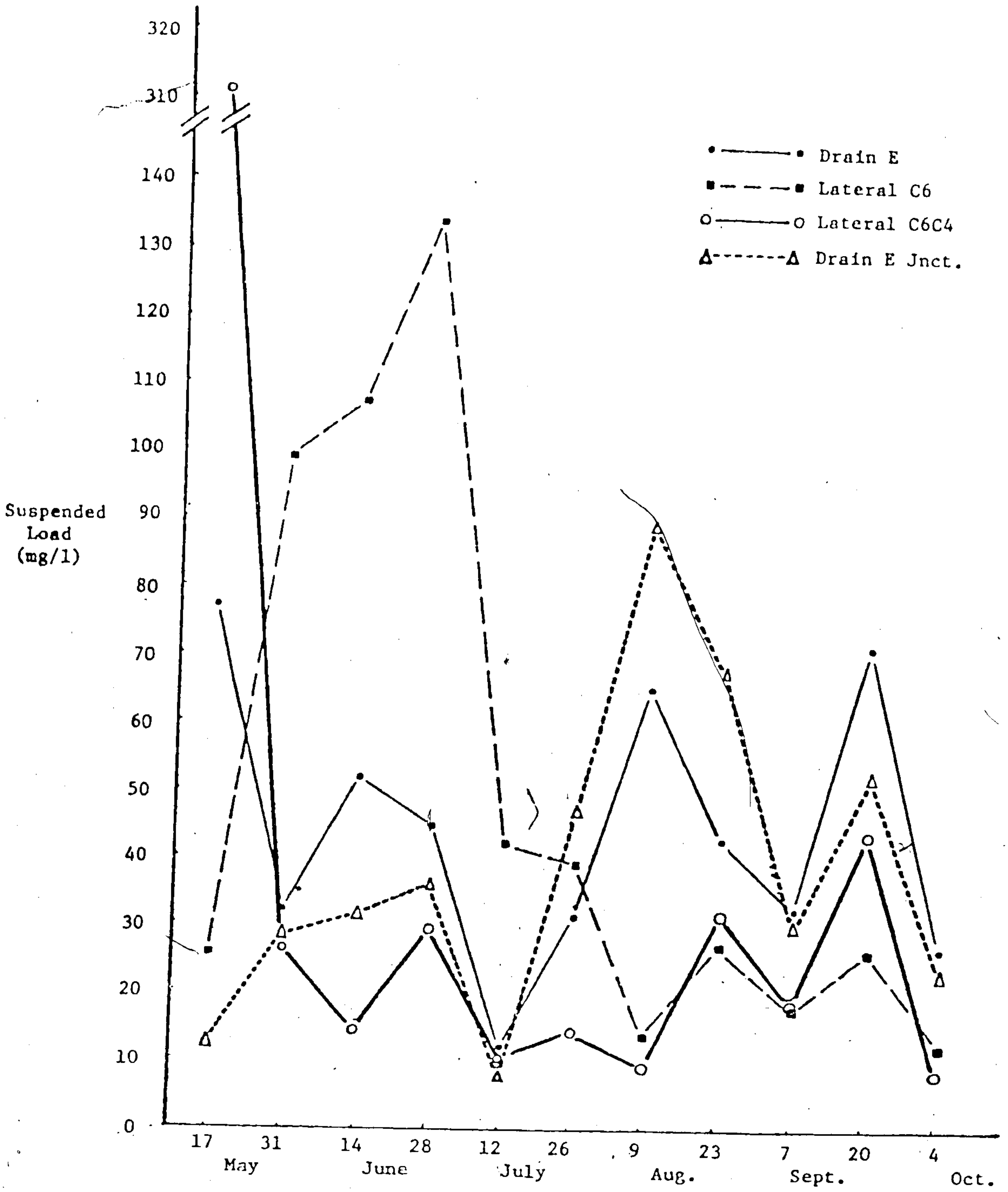


Figure 9. Sediment concentrations in the irrigation and drainage waters in the Vauxhall study area, 1976.

Table 20. Mean total and specific dissolved salt concentrations in the irrigation and drainage waters in the Hays area.

Site and Description	Total Dissolved Salts (mg/l)	Hardness (mg/l)	SAR	Na$^+$ (meq/l)	K$^+$* (meq/l)	Ca^{++} (meq/l)	Mg^{++} (meq/l)	Cl$^-$ (meq/l)	HCO$_3^-$ (meq/l)	CO$_3^=$ (meq/l)	SO$_4^=$ (meq/l)
1975											
Lateral R In (supply water)	260	140	0.97	1.1	--	1.5**	1.2**	0.17	2.4	0.0	1.6
Drain T-A (return flow)	283	160	0.98	1.2	--	1.4	1.7	0.19	2.7	0.0	2.3
Lateral R Out (a bypass water)	257	140	0.94	1.1	--	1.2	1.5	0.18	2.4	0.0	1.5
Drain 231 (a runoff water)	308	150	0.99	1.2	--	1.2	1.9	0.27	2.8	0.0	1.7
Drain C (bypass + runoff)	280	150	0.94	1.2	--	1.4	1.6	0.19	2.7	0.0	1.9
Cudrak Supply (a supply water)	246	140	0.95	1.1	--	1.5	1.2	0.20	3.3	0.0	0.2
Cudrak Drain (a runoff water)	273	140	0.96	1.2	--	1.6	1.3	0.31	3.0	0.0	1.7
1976											
Lateral R In (supply water)	260	140	0.92	1.1	0.06	1.3	1.5	0.12	2.7	0.0	1.6
Drain T-A (return flow)	300	160	1.02	1.3	0.11	1.4	1.8	0.14	3.3	0.0	2.0

Table 20. continued

Site and Description	Total Dissolved Salts (mg/l)	Hardness (mg/l)	SAR	Na$^+$ (meq/l)	K$^+$ (meq/l)	Ca^{++} (meq/l)	Mg^{++} (meq/l)	Cl$^-$ (meq/l)	HCO$_3^-$ (meq/l)	CO$_3^=$ (meq/l)	SO$_4^=$ (meq/l)
Lateral R Out (a bypass water)	248	140	0.95	1.1	0.06	1.2	1.5	0.12	2.5	0.0	1.6
Drain 231 (a runoff water)	344	180	0.95	1.4	0.16	1.6	1.9	0.13	3.2	0.0	2.7
Drain C (bypass + runoff)	273	150	0.97	1.2	0.09	1.4	1.6	0.13	2.8	0.0	1.7
Cudrak Supply (a supply water)	248	140	1.00	1.2	0.06	1.3	1.5	0.15	2.2	0.0	1.7
Cudrak Drain (a runoff water)	262	140	0.98	1.2	0.15	1.6	1.3	0.15	2.3	0.0	1.8
Cudrak Drain Downstream (a runoff water)	288	160	1.04	1.3	0.13	1.6	1.6	0.15	2.6	0.0	2.0

* No samples analyzed in 1975 for this ion. Three samples per site were analyzed in 1976 except for the Cudrak farm study sites.

** Estimated from Ca + Mg values according to the Ca:Mg ratio determined for each site in the 1976 analyses.

Table 21. Total seasonal salt loads in the irrigation and drainage waters in the Hays area for specific and total dissolved salts.

Site and Description	Total Dissolved Salts (t)	Na^{+} (t)	Ca^{++} (t)	Mg^{++} (t)	Cl^{-} (t)	HCO_3^{-} (t)	$SO_4^{=}$ (t)
<u>1975</u>							
Lateral R In (supply water)	6,300	630	730	350	150	3,570	1,700
Drain T-A (return flow)	2,000	200	200	150	60	1,130	780
Lateral R Out (a bypass water)	340	33	33	24	8.5	130	94
Drain 231 (a runoff water)	130	11	10	9.5	4.0	71	37
Drain C (bypass + runoff)	1,800	170	180	120	52	1,020	670
<u>1976</u>							
Lateral R In (supply water)	7,900	780	800	560	140	4,500	2,400
Drain T-A (return flow)	2,700	270	270	200	49	1,800	900

Table 21. (Continued)

Site and Description	Total Dissolved Salts (t)	Na^+ (t)	Ca^{++} (t)	Mg^{++} (t)	Cl^- (t)	HCO_3^- (t)	$SO_4^=$ (t)
Lateral R Out (a bypass water)	600	61	60	43	12	360	190
Drain 231 (a runoff water)	150	15	16	10	2.6	84	52
Drain C (bypass + runoff)	2,300	230	240	160	42	1,400	710

increase in total dissolved salts in the Drain T-A flow compared to that in the supply (significant at the 1 percent level). Though the differences were not always statistically significant, the concentration of salts in the return flow was consistently higher than that of the corresponding supply water; the average in 1975 and 1976 being 1.09 and 1.16 times greater, respectively.

The increases though small cannot be explained solely on the basis of diffusion of ions from the earthen conveyance structures or evaporation. The concentration in water monitored at Lateral R Out showed a slight decrease in total salts compared with that in Lateral R In flows even though the bypass water travelled some 18 km through earthen conveyance structures and had ample surface exposure to sun and wind. The increases in total dissolved salt concentration in the return flow from the Hays area was found to be in the same order as those at the Cudrak farm. The increase in total dissolved salt concentration in the farm runoff compared to supply water was 1.10 fold in 1975 and 1.06 fold in 1976, respectively.

On a weight basis much more salt was imported in the supply water than was exported in the return flow. Calculations for 1975 (in which the flow of supply water was estimated by the relationship: Supply Water = 3.3 x Return Flow, as found by Kerber (1975) for the Bow River Irrigation District) show a net import of 1.68 t/ha while for 1976 the net import amounted to 2.04 t/ha.

Comp[illegible] quality criteria in Table 7, the concentration of total disso[illegible]n the surface irrigation return flow from the Hays area would [illegible]se of the discharged water.

(b) Vauxhall area

Total dissolved salt concentrations for the Vauxhall area are given in Table 22, and corresponding salt loads are found in Table 23. Specific salt data are again included for convenience. Two t-tests, comparing the means of the total dissolved salt concentrations in water entering the area with those of water leaving via the Drain E return flow, showed no significant difference at the 5 percent level for either year. As in the Hays area, however, concentrations in the return flow always exceeded those in the supply water. The 1975 and 1976 increases were on the average 1.10 and 1.07 fold, respectively for sampling periods. On the last sampling trip there was no flow in Lateral C6 while some drainage water still flowed in Drain E -- about 0.01 m^3/sec. Consequently, Drain E was sampled while Lateral C6 was not. Including the salt concentration of 870 mg/l determined in this final sample increases the yearly mean from 306 mg/l to 377 mg/l for Drain E. The mean excluding this odd sample was used in calculating concentration increases.

Surprisingly, there was essentially no increase of total dissolved salts in the supply water as it passed through the area to the Lateral C6C4 monitoring site. The higher values for the salt concentration in the return flow is presumably attributable to subsurface drainage water that percolates to nearby surface drains, not to diffusion of ions from conveyance structures or evaporation. A small portion of the salt in the return flow may have the hog sewage lagoons as its origin. The apparently spectacular increase of 638 mg/l in the average total dissolved salt content of water passing the lagoons during 1975 was

Table 22. Mean total and specific dissolved salt concentrations in the irrigation and drainage waters in the Vauxhall area.

Site and Description	Total Dissolved Salts (mg/l)	Hardness (mg/l)	SAR	Na^+ (meq/l)	K^+ * (meq/l)	Ca^{++} (meq/l)	Mg^{++} (meq/l)	Cl^- (meq/l)	HCO_3^- (meq/l)	$CO_3^=$ (meq/l)	$SO_4^=$ (meq/l)
1975											
Lateral C6 (supply water)	279	150	0.81	1.0	--	1.7**	1.4**	0.19	2.8	0.0	1.9
Drain E (return flow)	306	224	0.87	1.3	--	2.6	1.9	0.19	3.4	0.0	3.0
Lateral C6C4 (a bypass water)	275	150	0.83	1.0	--	1.7	1.4	0.17	2.9	0.0	1.6
Drain E Jnct. (bypass + seepage)	327	180	0.90	1.2	--	1.7	2.0	0.21	3.0	0.0	2.4
Before Lagoons	589	160	0.83	1.1	--	1.8	1.4	0.37	2.8	0.0	2.7
After Lagoons	1230	250	1.80	3.2	--	2.2	2.7	1.60	3.9	0.0	4.5
1976											
Lateral C6 (supply water)	270	160	0.78	1.0	0.07	1.7	1.4	0.12	3.2	0.0	1.5
Drain E (return flow)	290	170	0.81	1.1	0.09	2.0	1.5	0.13	3.2	0.0	1.8
Lateral C6C4 (a bypass water)	272	160	0.80	1.0	0.08	1.7	1.4	0.11	3.1	0.0	1.6

Table 22. continued

Site and Description	Total Dissolved Salts (mg/l)	Hardness (mg/l)	SAR	Na^+ (meq/l)	K^+ (meq/l)	Ca^{++} (meq/l)	Mg^{++} (meq/l)	Cl^- (meq/l)	HCO_3^- (meq/l)	$CO_3^=$ (meq/l)	$SO_4^=$ (meq/l)
Drain E Jnct. (bypass + seepage)	305	180	0.83	1.1	0.09	2.1	1.6	0.13	3.3	0.0	1.9
Before Lagoons	272	160	0.78	1.0	0.07	1.8	1.4	0.12	3.0	0.0	1.5
After Lagoons	316	180	0.96	1.4	0.10	2.0	1.6	0.15	3.2	0.0	2.0

* No samples analyzed in 1975 for this ion. Three samples per site were analyzed in 1976.

** Estimated from Ca + Mg values according to the Ca:Mg ratio determined for each site in the 1976 analyses.

Table 23. Total seasonal salt loads in the irrigation and drainage waters in the Vauxhall area for specific and total dissolved salts.

Site and Description	Total Dissolved Salts (t)	Na^+ (t)	Ca^{++} (t)	Mg^{++} (t)	Cl^- (t)	HCO_3^- (t)	$SO_4^=$ (t)
1975							
Lateral C6 (supply water)	3,900	320	480	230	100	2,400	1,400
Drain E (return flow)	1,100	90	150	66	24	650	420
Lateral C6C4 (a bypass water)	550	47	67	33	13	340	170
Drain E Jnct. (bypass + seepage)	480	40	50	35	12	270	140
Before Lagoons	99	3.5	5.2	2.5	1.8	25	19
After Lagoons	210	10	6.2	4.7	7.8	120	30
1976							
Lateral C6 (supply water)	4,400	370	570	270	71	3,100	1,200

Table 23. (Continued)

Site and Description	Total Dissolved Salts (t)	Na^+ (t)	Ca^{++} (t)	Mg^{++} (t)	Cl^- (t)	HCO_3^- (t)	$SO_4^=$ (t)
Drain E (return flow)	1,410	120	200	90	23	960	420
Lateral C6C4 (a bypass water)	500	43	64	32	8.0	340	140
Drain E Jnct. (bypass + seepage)	860	73	120	54	14	520	260
Before Lagoons	180	15	24	12	3.1	120	51
After Lagoons	210	20	27	13	3.5	130	61

mainly a result of one sample taken at a time of negligible flow. The corresponding standard deviation of 1,446 mg/l for the After Lagoons samples indicates the variability in results. No samples were taken at negligible flow periods in 1976 and consequently the increases in concentration of 44 mg/l -- 1.16 times -- for that year represents more precisely the effect of the sewage ponds on the passing water. The mean was 316 mg/l and the standard deviation 60 mg/l for the After Lagoon site.

On the basis of a total salt budget (calculated as before), much more salt was imported into the Vauxhall study area via the supply water than was exported in the return flow. A net of 2.56 t/ha was imported during the 1975 irrigation season, while the 1976 net imports amounted to 2.76 t/ha.

Comparing the total dissolved salt concentrations in the surface return flow from the Vauxhall area with the criteria in Table 7, the return water can be considered to be of high quality except for the low flow period at the end of the 1975 season.

B. Hardness

Hardness, though not well defined, is an indirect measure of the capacity of water to form insoluble deposits in pipes, and supply polyvalent cations which precipitate soap from solution. It is expressed as mg/l of $CaCO_3$ and is sometimes referred to as "hardness as $CaCO_3$" or "calcium + magnesium hardness." According to the classification presented by Hem (1970), a hardness of 0 - 60 mg/l constitutes soft water; 61 - 120, moderately hard; 121 - 180, hard; and more than 180, very hard.

Except for the Drain E flow during 1975, all means of the supply and return flow water fall into the "hard" category. None of the values for the supply water to the two areas exceeded 180 mg/1. On an individual sample basis 6 of the 19 samples from Drain E exceeded 180 mg/1 of $CaCO_3$; 3 out of 18 samples at Drain T-A were "very hard". The bypass water in the Drain E basin showed almost no increase in hardness, but samples of the water flowing past the Drain E Jnct. site had hardness values exceeding 180 mg/1 11 out of 18 times. Since the Drain E Jnct. flow consists of bypass plus seepage water, the amount of subsurface drainage seems to be the controlling factor in the increase of hardness.

Compared to the water quality criteria in Table 7, all water samples exceeded the desirable hardness for municipal use -- even the supply water. In only one instance, the last sample from Drain E in 1975, did the hardness of any of the determined values exceed the maximum desirable level for industrial use.

C. Specific dissolved salts

The previously mentioned Tables 20, 21, 22, and 23 present data on specific dissolved salts, Tables 20 and 21 for the Hays area, Tables 22 and 23 for the Vauxhall area.

(a) Chloride

There were no samples taken for this study that had chloride concentrations in excess of limits in Table [illegible] except for one sample at the After Lagoons site. With the one exception, determined values for Cl^- were all less than 1 meq/1 (35.5 mg/1).

(b) Sulfate

Two samples taken during the study were polluted with respect to $SO_4^{=}$ content, that is they exceeded 5.2 meq/l. In 1975 when a flow was 0.003 m^3/sec at the After Lagoons site and in 1976 when a flow was 0.001 m^3/sec at the Drain 231 site, samples contained sulfate concentrations of 8.8 meq/l and 9.1 meq/l, respectively. Though the concentrations were high, the flows were low; hence total exports were low. Corresponding sulfate concentrations in the total return flows for corresponding sample periods were 1.2 meq/l and 1.8 meq/l. All other samples taken in the study had sulfate levels less than those in Table 7 (250 mg/l which equates to 5.2 meq/l) and would therefore be classed as having high quality with respect to sulfate content.

(c) Nitrate

Though discussed under the topic of eutrophication, nitrates may pose another problem -- infantile nitrate poisoning. However, there were no samples taken in the entire study which exceeded the maximum permissible level of 10 ppm NO_3-N. The closest concentration was 2.92 ppm in one sample of surface runoff from the Cudrak farm.

(d) Phosphates

Like nitrates, excessive phosphates can cause more problems than those associated with eutrophication. Water with total-P concentrations in excess of 0.100 ppm inhibit the flocculation process in water treatment. Turbidity due to sediment is therefore more difficult to reduce in high phosphate waters, and subsequent cost of water treatment is increased.

In the irrigation return flow from both the Hays and Vauxhall areas, the determined total-P concentrations exceeded the maximum desirable level of 0.100 ppm once each year, namely the first sample period in each study year.

Means averaged 0.065 ppm and 0.039 ppm for the Hays and Vauxhall areas, respectively. These values are surprisingly low for the flood-irrigated area since total-P in runoff from the study farm in that area exceeded 0.100 ppm in nearly every sample. Factors causing low total-P concentrations in the return flow were discussed in the section on eutrophication.

(e) Sodium

High sodium concentrations relative to calcium plus magnesium (high SAR's) in irrigation waters result in a decrease in the stability of soil aggregates of the receiving lands. According to the United States Salinity Laboratory Staff (1954), use of waters having SAR values of 4 and above present a very high potential for the alkalization of soils; 3 to 4 a high potential; 2 to 3 a medium potential; and 1 a low potential. The maximum permissible SAR's of 8 to 18 in Table 7 are those that directly affect plant growth adversely.

The highest mean SAR of the return flow was 1.02, determined the Drain T-A area in 1976. SAR values in runoff from the study farm in that area were less than 1.0 in both years. For one low flow in 1975, Drain E had one SAR value in excess of 1.0, but all other samplings in both years at that site were less.

Average increase in the sodium concentration of waters discharging into the rivers compared to supply water was 5 percent for the

Hays watershed, and 20 percent for the Vauxhall area. Nevertheless, with respect to sodium content, the surface return flow from the study areas can be said to be of high quality.

(f) Carbonate and bicarbonate

Because of the precipitation of $CaCO_3$ from solution, SAR values of the soil solution may be increased. Waters high in bicarbonate and carbonate ions have been found to cause alkali conditions in receiving soils. As recorded in Table 7, the maximum permissible level of residual Na_2CO_3 in irrigation waters is 1.25 meq/l. Residual Na_2CO_3 is defined as $(HCO_3^- + CO_3^=) - (Ca^{++} + Mg^{++})$, where concentrations are in meq/l.

Of all the sites monitored in both the Hays and Vauxhall areas, only two gave positive values for residual Na_2CO_3. These were 0.06 meq/l for Lateral C6 in 1976 and 0.04 for Drain T-A in 1976. There was no carbonate found in any of the analyses.

Essentially, there is no residual Na_2CO_3 in the irrigation or return flow waters from the Hays and Vauxhall study areas.

Alkalinity

Combining the various criteria in Table 7, a range in pH of 7.0 to 8.3 is acceptable for all stated water uses. Most pH values in the study fall between 7.0 to 8.5. The lowest value in the study, 7.0, was determined in the surface runoff from the Cudrak farm; the highest, 9.2, was determined on a water sample from the Lateral R In site. High pH values in the Drain T-A waters corresponded with high values in the supply water. Runoff from the Cudrak farm and from the nine farms contributing to Drain 231 showed consistently low values relative to the supply water, indicating that the contact between the applied water and the irrigated land lowered high pH's.

The pH of the return flow water from both study areas was frequently in excess of maximum desirable levels for recreational, municipal, industrial, and agricultural use. However, high pH values in the return flow corresponded with high values in the supply water. The application of this alkaline irrigation water and subsequent contact with the land resulted in improved water quality with respect to pH.

Temperature

Measured water temperatures for Drain T-A and Drain E never exceeded the 30°C maximum stipulated in Table 7. This was the case for all monitored sites -- even the surface runoff from the Cudrak farm. Water temperature of surface runoff corresponded closely to air temperatures with expected lags due to the differing thermal capacities of air and water. Temperatures below the 13°C minimum for irrigation were reached in the month of September for most sites, but little water is applied to crops at this time. Most irrigation in September is for supplementing winter precipitation to ensure adequate moisture in the spring.

Water temperature was often positively and significantly correlated with flow volume. This indicates that maximum return flows occur at maximum air temperatures, more correctly, at maximum periods of water demand.

Some Important Factors Affecting Return Flow Quality

A. Flow volume

Though not specifically stated, it has been implied in previous sections that at low flows the concentration of dissolved constituents in the surface drainage water tend to be relatively high, while at higher flows concentrations decrease. This is based on the premise that the supply of constituents affecting quality remains constant for any total flow. Dilution then, is the controlling factor in the resultant concentration. Conversely, suspended load tends to increase with increases in flow volume and velocity. Of course these are generalizations. For example, an increase in the concentration of salinity constituents in the bypass water relative to the supply water would be expected because the much lower flows encountered in the bypass water would increase the surface area contacts between water and bed surface, and water and evaporative forces per unit volume of flow. This was not the case as mentioned previously. Rather, the concentrations in the bypasses remained relatively constant from the points of supply to the points of spillage.

On the other hand, the above premise is well supported by some of the correlation coefficients in Table 24, most noticeably for the Cudrak farm study in 1976.

Correlations of salinity, nutrients, and suspended load with flow were often significant at the 5 percent level. EC x 10^6, SAR, Na^+, $Ca^{++} + Mg^{++}$, Mg^{++}, Ca^{++}, K^+, $SO_4^=$, HCO_3^-, Cl^-, NO_3-N, and ortho-P were all negatively correlated; that is, the more water flowing off the field, the more dilute the concentrations. Suspended load and total-P both

Table 24. Correlation coefficients, "r", for linear correlation of flow with studied components of water quality at a few sites.

	Drain T-A		Cudrak Drain		Drain E	
Component	1975	1976	1975	1976	1975	1976
EC x 10^6	-0.24	-0.66*	-0.51	-0.93**	-0.67	-0.67*
SAR	0.81**	-0.33	-0.13	-0.75*	-0.54	-0.50
pH	0.73**	-0.01	0.30	0.33	-0.38	0.43
Na^+	0.34	-0.65*	-0.17	-0.91**	-0.68	-0.57
K^+	--	-0.99**	--	-0.85**	--	-0.53
$Ca^{++} + Mg^{++}$	-0.67	-0.35	-0.06	-0.61	-0.73*	-0.61*
Cl^-	0.77*	0.27	-0.32	-0.29	-0.05	-0.09
HCO_3^-	-0.67	-0.59	-0.70*	-0.22	-0.82*	-0.18
$SO_4^=$	-0.13	-0.23	-0.21	-0.64	-0.52	-0.56
NO_3-N	0.68	-0.40	-0.73**	-0.70*	0.04	-0.53
Total-P by persulfate	0.23	-0.51	-0.04	0.46	0.07	-0.47
Total-P by perchloric	--	-0.01	--	0.72*	--	-0.11
Ortho-P	0.28	0.03	0.23	-0.37	0.77*	-0.51
Suspended load	--	0.02	--	0.69*	--	0.29
Water temperature	0.40	0.99**	--	--	0.34	0.40

* Significant at the 5 percent level.

** Significant at the 1 percent level.

increased with increasing flows. These findings match the previously discussed relationship of flow and quality constituent concentrations. The rest of the data in the table, though usually positively or negatively correlated where expected, refute the idea that flow is the main controller of ion and sediment concentrations in the return flow; correlations of quality with flow are rarely significant. Because of the lack of significance, it is obvious that there are other important factors affecting water quality besides flow.

B. Inherent quality of the supply water

The most important factor in determining the quality of the return flow from the study areas is the concentration of components affecting quality in the supply water. As discussed throughout other sections, there is only a small increase in concentrations over-and-above those in the supply water. Thus, the quality of the return flow is closely related to the quality of the supply water.

From the original point of contact with the earth, the supply water has equilibrated with transported sediments and channel beds of small creeks, streams, rivers, reservoirs, and canals. Results obtained for bypass water quality in both areas during both years lead to the observation that the transport of the supply water through an additional few kilometers of lateral results in essentially no change in the total dissolved salt concentration of that water -- regardless of flow. Therefore, the quality of that portion of the total return flow which has bypass origin, is not correlated with flow but rather with the inherent quality of the supply water.

C. The relative proportions of subsurface drainage, surface runoff, and bypass water

Subsurface and surface drainage water have a much greater water-soil interface than does the bypass water in a hydraulically efficient lateral, and consequently can be expected to have higher dissolved salt concentrations. The time of contact between soil and water is maximized in the case of subsurface flow. This can be illustrated by the total dissolved salt concentrations of the three surface return flow components above supply water concentration: 600 mg/l for subsurface flow (as determined in the "drain out" period for Drain E when no supply water entered the area in October), about 20 mg/l for surface runoff (as determined in the Cudrak farm study), and 0 mg/l for bypass waters (as determined in Lateral C6C4 and Lateral R Out both years).

The data in Tables 25 and 26 are indicative of the relative contributions actual and estimated, from these three sources to the total flow and to the salt content of the return flow. The salt concentrations of subsurface drainage, surface runoff, and bypass water, when weighted by the flow volumes, result in total dissolved salt increases of 21 mg/l and 22 mg/l for Drain E, 1975 and 1976, respectively; and 38 mg/l and 40 mg/l in Drain T-A for the two years. Corresponding measured increases over supply water concentrations were 27, 20, 23, and 40 mg/l, respectively -- very close to the estimated ones. Without subsurface flows the estimated concentration increases would equal 0.1, 0.1, 7, and 11 mg/l -- far from the measured ones. Though subsurface flows are small, it would appear that most of the salt increase in the return water over supply water concentrations is attributable to this source.

Table 25. Volumes of subsurface flow, surface runoff, and bypass water in the Hays and Vauxhall areas.

Year	Subsurface Flow* (ha-m)	Surface Runoff** (ha-m)	Bypass Water*** (ha-m)
<u>Hays</u>			
1975	30	193	[illegible]
1976	39	422	3
<u>Vauxhall</u>			
1975	9.0	1.2	249
1976	12	1.2	325

* Estimated from base flows measured at the end of the irrigation seasons when no water was delivered to the study areas.

** Hays area -- estimated from the relationship surface runoff = 0.3 x delivery water, as determined for 9 farms contributing to Drain 231.

Vauxhall area -- estimated from point measurements at 2 of the 3 flood-irrigated fields in the study area.

*** Measured.

Table 26. Calculated and measured total dissolved salts in the Drain T-A and Drain E return flows.

Year	Drain T-A Calculated Concentration (mg/l)	Drain T-A Measured Concentration (mg/l)	Drain E Calculated Concentration (mg/l)	Drain E Measured Concentration (mg/l)
Subsurface flow + surface runoff + bypass water				
1975	298	283	300	306
1976	300	300	292	290
Surface runoff + bypass water				
1975	267	--*	279	--
1976	271	--	270	--

* Since the return flow included subsurface flow, there was no way of measuring salt concentrations in surface runoff plus bypass water alone.

D. Proximity to a nutrient source

As discussed in the section on eutrophication, the nutrient status of the runoff water from the Cudrak farm changed dramatically as it passed through a 2.5 km stretch of drain. In 1975, downstream flow from the farm was not sampled. However, it was noted in that year that the nutrient status of the Drain T-A discharge did not reflect the surface runoff quality. The 1976 sampling revealed a drop in the nitrate concentration to 1/10 the original level over the 2.5 km stretch, and a drop to 1/2 the original total-P concentration. The question as to what biological, chemical, and/or physical processes were responsible for this change was not totally answered by this study, but the following facts can serve as guides for further investigation: (1) the aquatic vegetation in the drain required nutrients for growth; (2) NO_3-N can be reduced to NH_4-N in the anaerobic portion of channel beds; and (3) suspended phosphorus-bearing sediments settle with time. These factors may be important in demonstrating why, on a regional basis, the nutrient status of water bodies are rarely correlated with fertilizer use on nearby agricultural land. A logical hypothesis with regard to nutrients may be implied from the data: the farther with respect to both time and space that nutrients travel from their source, the greater will be the fluctuations in the nutrient status of the transporting water.

Changes in Receiving Stream Quality as Affected by Return Flow

Previous to this section the return flow quality has been judged on the basis that this water is to be the direct source of supply for various uses. However, the irrigation return flow from the study sites discharges into two rivers and subsequent dilution occurs. The phrase "dilution is not the solution to pollution" conveys the reasoning for assessing the return flow on its own merit rather than making return-flow-plus-river quality the primary consideration. The quality of the receiving stream is an important consideration in return flow studies, however, and cannot be overlooked.

Table 27, unpublished data from Dr. Oosterveld's study, give mean values of river quality data consisting of four samples at each site. Corresponding 1976 data for the study drains are also given for comparison, as are the concentrations of constituents in river water that would result from the additions of the return flow to the river. Due to the high flow in the rivers relative to the drains, the effect of the irrigation return flow quality on river water is in essence negligible. If concentrations in the return water were higher, or the return flow sites more numerous, then discharged nutrients, salts and sediments could impair water quality in the rivers involved.

Soil

A Donor of Pollutants

Setting aside all other considerations and estimating the quality of the return flow from the study areas on the basis of concentrations of nutrients and salts in surrounding soils, the return flow from the Vauxhall area should have had a phosphorus content in the order of 2.2 times that of the Hays return flow, a nitrate content 3.8 times,

Table 27. Water quality in Drain T-A, Drain E, the Oldman River, and the Bow River, and corresponding flow-weighted concentrations in river-plus-return-flow water, 1976.

Water Quality Component	Drain T-A	Oldman River	Drain T-A plus Oldman River	Drain E	Bow River	Drain E plus Bow River
pH	8.6	8.0	8.0	8.5	7.7	7.7
SAR	1.0	0.41	0.41	0.8	0.37	0.37
TDS* (mg/l)	300	197	197	290	207	207
Hardness (mg/l)	160	130	130	175	138	138
Na^+ (meq/l)	1.3	0.47	0.47	1.1	0.43	0.43
$Ca^{++} + Mg^{++}$ (meq/l)	3.2	2.59	2.59	3.5	2.75	2.75
Cl^- (meq/l)	0.14	0.06	0.06	0.13	0.03	0.08
HCO_3^- (meq/l)	3.3	2.57	2.57	3.2	2.33	2.33
$SO_4^=$ (meq/l)	2.0	0.92	0.92	1.8	0.97	0.97
NO_3-N (ppm)	0.19	0.20	0.20	0.28	0.42	0.42
Total-P** (ppm)	0.061	0.072	0.072	0.076	0.158	0.158
Ortho-P (ppm)	0.010	0.007	0.007	0.027	0.028	0.028
Suspended Load (mg/l)	34	29.8	29.8	44	15.9	16.0

* Denotes total dissolved salts.

** Total-P by the persulfate method.

and a salt content 1.8 times greater. Estimating the return flow quality on the premise that all the fertilizer that was applied in the Hays area entered the return flow over the growing season, NO_3-N concentrations discharged from the study basin to the river would be about 10 ppm in the return flow, and P levels about 5 ppm. The corresponding N and P values for Vauxhall would be 10 ppm and 8 ppm, respectively. Actual values were not even near these "predicted" ones. Though the previous calculations are a bit absurd, they do illustrate a key point: though potential pollutants are contained within a soil, they are not necessarily transported to a neighboring water body. The soil is a reservoir of pollutants, but transport to a receiving water body is necessary before pollution will occur. The transport of potential pollutants by diffusion, surface runoff, subsurface flow, and soil drifting have been discussed in previous sections. Subsurface flow with a concentration between 800 and 900 ppm seemed the major contributor of dissolved salts to the return flow in this study; erosion by surface flow and by wind were responsible for major increases in suspended loads in the return flows from the study areas; and all three contributed to nutrient concentrations in the drains.

A Receiver of Pollutants

A. Salinization

(a) An attempted salt balance study

In the proposal of project objectives formulated in 1975, it was decided that a soil salt balance should be attempted. The goals were to determine whether any change in salt content due to the two seasons of irrigation could be detected, and to relate such changes to quality of irrigation water and return flow. The results, as will be discussed in more detail shortly, show this was not feasible. Salt concentrations were consistently lower in the 1976 fall sampling than in the 1975 spring sampling in 78% of the samples. The t-tests applied showed significant decreases (significant at the 1 percent level) for the three sampling depths in the top 90 cm of the Hays area. The level of significance never exceeded the 7 percent level for any of the sampling depths in the Vauxhall area. On the basis of these findings, it is suggested that the time of year, in particular with respect to the irrigation season, is very important in scheduling soil sampling for salt balance studies. Yearly samples should be taken at the same time period each year of the study, and samples taken during the year would generate data which could increase understanding of processes affecting the studied salts.

(b) Salt imports and leaching losses and their potential effect on plant growth

Discussion of some information from the salt balance is warranted, however. The net salt imports of about 1.7 and 2.6 t/ha/yr into the Hays and Vauxhall areas, respectively, are definite threats to crop production unless the salts are removed from the rootzone by some means. Assuming no loss of the salts added in the irrigation water, the soil salinity of a 120 cm rootzone would be increased 100 and 150 ppm/yr in the study areas. Using the nomograph given by the United States Salinity Laboratory Staff (1954, p. 17) which relates the percent salt in a soil (having a known saturation percentage) to the osmotic pressure of the saturation extract, the osmotic pressure caused by the added salt can be estimated. For the average Hays soil, after a salt addition of 100 ppm for 10 years, the increased salinity in the top 30 cm would exert an osmotic tension of 3.54 atmospheres at field capacity. Extrapolating from a plot of pressure versus water content (a soil moisture curve) a reduced plant available moisture level can be determined for the estimated osmotic pressure. This osmotic pressure, 3.54 atmospheres, would effectively reduce the available moisture 74.6 percent in the top 30 cm of soil in the Hays area. For the rest of the rootzone, the decrease would be about 65 percent. In Vauxhall, the salt additions of 150 ppm/yr to the rootzone would inhibit plant uptake of about 80 percent of the moisture in the top 30 cm of soil and about 76 percent in the remaining 90 cm after 10 years. On this basis, after the 56 years of irrigation at Vauxhall and about 25 years at Hays, there should be little crop growth in either area.

However, this is definitely not the case. In fact, the opposite seems true -- irrigation appears to have reduced the salt content of the soil between spring, 1975, and fall, 1976. The 1975 and 1976 sampling showed a large decrease in salts between spring of 1975 and fall of 1976 (Table 28). Figure 10 depicts the percentage of salt content decreased over the two irrigation seasons. From the determined electrical conductivity values, 10 t/ha of salt was leached from the 240 cm depth in the Hays area; from the Vauxhall area there was a net loss of 6.1 t/ha from the same depth. At this rate of salt loss, the previous years of irrigation would have removed almost all the soluble salts from a 240 cm depth in the two areas. This obviously is not true as seen in the spring and fall sample analyses, but the previously discussed data do show that there is a favourable salt balance with respect to plant

Table 28. Mean soluble salt content with depth in the Hays and Vauxhall soils and underlying parent materials in the spring, 1975, and fall, 1976.

	Number of Samples	EC × 10^3	SAR	Na^+ (meq/l)	K^+ (meq/l)	Ca^{++} (meq/l)	Mg^{++} (meq/l)	Cl^- (meq/l)	HCO_3^- (meq/l)	$SO_4^=$ (meq/l)
Hays										
0-30 cm										
1975	19	2.12	1.4	4.4	5.6	14	9.5	0.89	12	13
1976	19	0.91	1.3	2.0	3.4	4.1	3.2	0.47	5.0	3.5
30-60 cm										
1975	18	1.61	1.9	4.9	4.9	8.9	5.8	0.80	8.4	7.8
1976	18	0.66	1.7	2.6	1.6	2.5	2.2	0.97	3.4	3.0
60-90 cm										
1975	18	1.92	2.9	6.7	4.1	8.7	7.7	0.77	6.6	15
1976	18	0.82	2.3	4.2	1.5	2.0	2.6	0.51	3.2	5.0
90-180 cm										
1975	18	4.04	3.7	14	3.8	21	20	0.81	4.1	52
1976	18	3.15	3.0	12	3.6	15	18	0.90	2.3	39
180-240 cm										
1975	7	4.90	4.3	22	2.9	22	25	0.79	3.1	67
1976	6	4.39	2.9	15	5.9	25	24	0.42	2.4	60

Table 28. continued

	Number of Samples	EC $\times 10^3$	SAR	Na^+ (meq/l)	K^+ (meq/l)	Ca^{++} (meq/l)	Mg^{++} (meq/l)	Cl^- (meq/l)	HCO_3^- (meq/l)	$SO_4^=$ (meq/l)
Vauxhall										
0-30 cm										
1975	16	3.92	2.4	13	5.0	29	17	0.80	13	43
1976	14	2.11	2.4	9.0	3.7	11	9.5	0.97	4.0	24
30-60 cm										
1975	16	4.76	3.6	20	5.4	30	21	0.83	8.2	60
1976	14	3.08	3.5	17	2.1	13	16	0.99	3.0	38
60-90 cm										
1975	16	5.27	6.1	29	4.7	26	22	1.8	5.2	72
1976	14	4.49	6.2	31	3.0	14	25	2.2	2.5	59
90-180 cm										
1975	16	5.42	7.2	33	4.2	18	21	2.1	4.0	69
1976	14	6.83	7.5	43	5.4	21	36	3.3	1.8	92
180-240 cm										
1975	4	7.07	8.4	47	3.9	19	31	5.0	3.7	93
1976	4	5.28	5.8	29	6.0	20	25	2.1	1.8	67

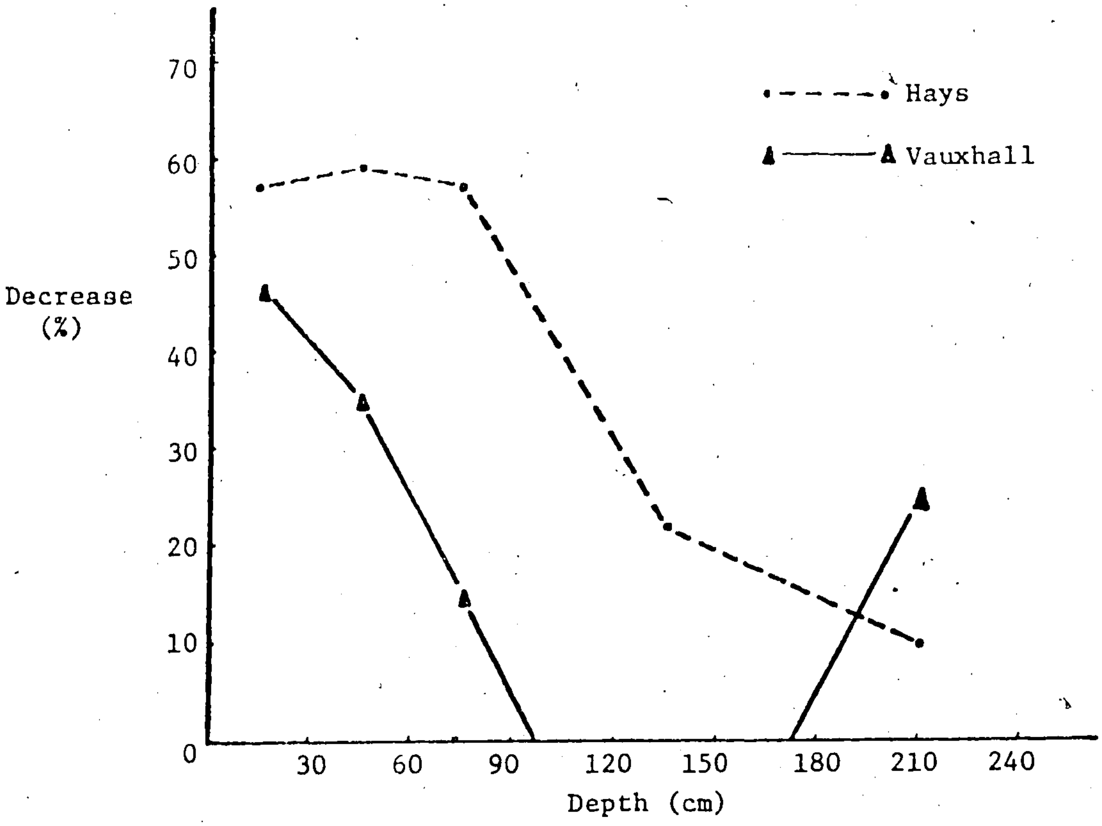


Figure 10. Decrease in the soluble salt content of the studied Hays and Vauxhall soils from the spring, 1975, to the fall, 1976.

growth between spring, the time before salt imports in the irrigation water, and fall, the end of seasonal salt imports.

Though not studied in detail, some leaching of soluble salts from the upper soil horizons may occur in early spring when soil moisture levels are high, evapotranspiration minimal, and rainfall plentiful. Of the 13 cm of precipitation recorded in the Hays area during 1975 in the months of June through August, 8 cm fell during June, 7 cm of which came in one week. The Vauxhall area received 8 cm of precipitation that same week. Considering the 1975 soil moisture levels as determined in the latter part of May and the water holding capacities of the Vauxhall and Hays soils, the June precipitation could have raised soil moisture contents above 1/3 bar percentages in both the Ap and Bm horizons of the study areas. The resultant gravitational water, 1 cm in the Hays area and 2 cm in the Vauxhall area, would rapidly percolate to the C horizon where it would be retained. The Cca horizon in both areas could hold 20 cm more water before reaching field capacity (at the time of the May soil sampling). Thus, rainfall may account for some of the leaching of salts from the upper portions of soil profiles in the study areas.

Explaining why the soil salt content in the spring of 1975 could be at the determined level is another matter. The author would conjecture that salts move up from the ground water to the soil profile during the period of late fall to early spring. The following statement from van Schaik and Rapp (1970) adds credence to the hypothesis: "In irrigated areas, the translocation of water during the winter from shallow water tables to the freezing zone is important. For example, thermally induced moisture flow may transfer salts upward and salinize the surface soil." Groundwater in the Vauxhall area contains about 7,000 ppm dissolved salt (unpublished data, Lethbridge Research Station). The deposition of salts from 8.7 cm of ground water into the Vauxhall soils would raise the soluble salt content in the 240 cm depth from the fall (1976) to the spring (1975) level. Similarly, 14.3 cm of ground water contain as much salt as that lost from a 240 cm depth in the Hays area over the 1975 to 1976 sampling period.

(c) Salinity status of the Hays and Vauxhall soils

Some saline soils were found in the study areas. Table 29 gives the number of samples in which the electrical conductivity values of the saturation extract exceeded 4.0 mmhos/cm. Both 1975 and 1976 sampling periods were included in the table. The 1976 frequencies were slightly less than those in 1975. Noticeably, the salt content of the Hays soils to a depth of 90 cm rarely exceeded the 4.0 mmhos/cm division between saline and non-saline soils. At the sampling depth of 90 to 180 cm the incidence of high salt content increased sharply rising from less than 10 percent of the samples in the 60 to 90 cm depth to greater than 40 percent of the samples in the 90 to 180 cm depth. At a depth of 180 to 240 cm, 64 percent of the soil samples were saline.

A greater frequency of saline soils was found in the Vauxhall area. Of particular concern is the observation that over 25 percent of the sampling sites in this area had saline rootzones. These high salt contents would seem critical, for this area has an average water holding capacity of only 19 percent of which just 11 percent is available for plant use. The practice of continually operating sprinkler systems during the irrigation season, except for short periods, is perhaps well advised. These frequent irrigations as well as the medium to high salt tolerance of the major crops in the area, wheat, potatoes, barley and

Table 29. Frequency of electrical conductivity values in excess of 4 mmhos/cm for the Hays and Vauxhall soils.

Depth (cm)	Hays		Vauxhall	
	Occurrence	% Occurrence	Occurrence	% Occurrence
0- 30	1/37*	3	8/30	27
30- 60	1/37	3	9/30	30
60- 90	3/37	8	14/30	47
90-180	16/37	43	21/30	70
180-240	9/14	64	6/8	75

* For example, 1/37 means that of 37 samples, 1 had an EC x 10^3 value greater than 4.0 mmhos/cm.

sugar beets, may account for what appears to be excellent crop growth there. Unfortunately, no yield information was sought in the farm interviews.

Several factors may contribute to the difference in salt content of the two areas. First, there was a greater water application in the Hays area where an average of 80 ha-cm of water was delivered to the farms during the two seasons, while a total of 73 ha-cm of water was delivered to Vauxhall farms for the corresponding time period. Secondly, the hydraulic conductivity is greater in the Hays soils for all pedogenic divisions, though water movement in the Vauxhall soils should not be restricted. The lowest mean value of hydraulic conductivities determined for the Vauxhall area was 2.7 cm/hr for the Ck horizon. Hydraulic conductivities in the till were higher. A third possible cause is the greater depth to the water table in the Hays area. van Schaik and Rapp (1970) found the water table at a site in the Vauxhall study area to recede to about 220 cm in winter months. Irr tion raised the water table to within 60 cm of the soil surface. Fr a map of depth to water table in the Hays region (Lethbridge Research Station), the water table in the Hays study area lies below a 300 cm depth. The study of the effect of the above three factors on soil salinity levels was not part of this project and will not be discussed further.

B. Alkalization

Few sample sites in either area had exchangeable-sodium-percentages in excess of 15 at any depth. From the data in Table 30, it may be inferred that there is little problem of soil alkalization in the study areas. This condition is probably a reflection of the high quality of irrigation water imported into this region; SAR values were usually less than 1.0 and residual sodium bicarbonate content was essentially nil.

(Note: Raw data, including flow data and chemical and physical analyses, are on file at the Agriculture Research Station, Lethbridge, Alberta.)

Table 30. Frequency of exchangeable-sodium-percentages in excess of 15 for the Hays and Vauxhall areas.

Depth (cm)	Hays Occurrence	Hays % Occurrence	Vauxhall Occurrence	Vauxhall % Occurrence
0- 30	0/18*	0	0/16	0
30- 60	0/18	0	1/16	6
60- 90	0/18	0	2/16	13
90-180	0/18	0	2/16	13
180-240	1/7	14	1/4	25

* For example, 0/18 means that of 18 samples, 0 had an exchangeable-sodium-percentage greater than 15.

SUMMARY AND CONCLUSIONS

Irrigation return flow has the potential to contribute substantial quantities of pollutants to natural water bodies. At polluting levels, nutrients, total and specific dissolved salts, and sediments impair recreational, municipal, industrial, and agricultural use of water. Discharges of return flow to natural water bodies may also cause shifts in the aquatic ecosystem of the receiving water. Algal blooms and fish kills are two major concerns.

To determine the quantities and sources of water, nutrients, salts, and sediments discharged to the Bow and Oldman Rivers from parts of the Bow River Irrigation District, two watersheds were monitored at key locations as to surface water quantity and quality during the 1975 and 1976 irrigation seasons. One watershed, the Drain T-A or Hays area, consisting of 2,560 cultivated hectares, is flood-irrigated and discharges return flow to the Oldman River. The other 1,090 hectare watershed, Drain E or the Vauxhall area, is sprinkler-irrigated. Return flow from this latter area discharges into the Bow River.

It was expected that nutrient levels would be significantly increased in the return flow, particularly the phosphorus content of the flood-irrigated area and the nitrate content of the sprinkler-irrigated area. The total-P and NO_3-N concentrations of surface runoff downstream from a study farm decreased substantially over a 2.5 km distance from the source. The P decrease to one-half the original level was attributed to a 55 percent reduction in suspended load due to settling of the previously eroded soil particles. This portion of the study helped to explain the low nutrient levels found in the return flow from the two areas. Phosphorus discharges amounted to <0.2 kg/ha/yr from the Hays area; <0.3 kg/ha/yr was discharged in the surface return flow from the Vauxhall tract. With respect to the amount of algae that could be produced from these P discharges per cubic meter of receiving water, the Hays discharge can be equated to an algal biomass of about 0.05 g/m^3 and the Vauxhall P discharge can be equated to 0.04 g/m^3. The corresponding weights of NO_3-N were <0.6 kg/ha/yr from the Hays area and <1.03 kg/ha/yr from the Vauxhall area. These values can be equated to an algal biomass equivalence of about 0.015 g/m^3 in the Oldman River, and about 0.014 g/m^3 in the Bow River. Though a group of return flow sites may raise nutrient levels of the receiving streams to adversely eutrophic levels, the discharges from these two areas alone would not be classed as polluting.

Surprisingly, the highest mean sediment concentration, other than in the surface runoff collected as it ran off a field in the study farm, was found in the return flow from the sprinkler-irrigated area, not from the flood-irrigated area. This resulted from a deposit of windblown topsoil which had accumulated in the eastern channels of the Vauxhall area. Gentle channel bed slopes in the Hays area, whose distribution and drainage network is on contours, result in the sedimentation of much of the suspended material derived from farm runoff. The suspended loads for the Hays and Vauxhall areas were respectively 33.5 and 43.5 mg/l on the average. At these concentrations aquatic organisms, particularly sport fish, are subjected to a moderately high level of protection.

Total dissolved salt content of the return flow was not increased substantially over that of the supply water, averaging 27.5mg/l for all site-years. Salt increases were slightly higher in the Hays area. The majority of salt increase in both areas can be attributed to a small amount of subsurface flow which had a total dissolved content of about 800 to 900 mg/l. Farm runoff, having passed over soil surface, had only slightly higher dissolved salt content than corresponding applied water; an average of 22.5 mg/l was determined for two studies at a farm site in Hays. No increase in the total dissolved salt content of bypass water occurred in either area either year. These low salt increases as well as inherent good quality of the supply water when compared to quality criteria lead to the conclusion that the total dissolved content of the return flow would not impair downstream use of that water.

Both supply and return flow water in the study areas are hard. Hardness was always increased in the return flow, particularly in the Drain E waters. Though no problem to industry, some municipal users would prefer softer water.

Chloride, sulfate, nitrate, phosphate, SAR, carbonate and bicarbonate levels were all found at acceptable levels for municipal, industrial and agricultural uses of water.

The pH values determined in the return flow samples were frequently in excess of acceptable levels. High pH in the return flow corresponded with high levels in the supply water and was usually less, thus representing an improvement in quality.

Water temperatures were positively correlated with flow volume. This indicates that peak return flows occurred at times of high air temperatures which in turn corresponded to times of high water demands. Water temperatures were not increased over the maximum limit for use.

Considering the dilution of the return flows by the receiving waters, there were essentially no increases in the nutrient, dissolved salt, or suspended load concentrations in the Bow and Oldman Rivers as a result of the discharge from the study areas.

Because of the vast quantities of potential pollutants held within the soil, agriculture has often been cited as a major source of pollutants into natural waters. This study has shown that though concentrations are high in the soil, pollutant concentrations in return flow are not necessarily so. The transport of pollutants from the soil to a water body is necessary before pollution will occur. Leaching can move substantial amounts of dissolved salts, including nitrates, to a neighboring water body. Surface runoff and soil drifting can transport both substantial quantities of sediment and associated phosphorus to a drainage channel discharging into a natural sink. The extent of this transport in the surface return flow from the Drain T-A and Drain E study areas seems to be minimal.

A study of salt balance in the soil between the spring of 1975 and the fall of 1976 failed its original intent. Rather than being able to account for the increase in salt imported in the irrigation and rain waters, a substantial decrease in soluble salts was noted for that time period in a 240 cm depth. These decreases amounted to 10 t/ha from the Hays area and 6 t/ha from the Vauxhall area over the two irrigation seasons. This information helps explain why the soils in these areas, though they receive salt imports of about 1.7 to 2.8 t/ha/yr, generally do not have excessively saline rootzones. Ground water having a concentration of about 7,000 ppm may replenish much of the leached salts during the non-irrigation season.

Though the Hays area was not found to have any extensive salt problems, concern is expressed over the findings in the Vauxhall area. About 25 percent of this area does have a soluble salt content that would impair growth of sensitive plants. The practices of maintaining high soil moisture levels and growing less sensitive crops are factors that have lead to successful agricultural endeavors in this area.

ESP values rarely exceeded 15 percent in either area in the sampled 240 cm depth. This was attributed to the low SAR' of the incoming irrigation water. The problems associated with alkalization should not be as evident as the problems associated with salinization in these two areas.

Pertaining to this study, one important conclusion about the employed research procedures was reached: salt balance studies of an irrigated area must include soil samples taken at the same time of year, more accurately at the same time relative to the irrigation season.

This study has raised several interesting questions. Research to answer the following would be beneficial:

1. What processes are responsible for the large decrease in NO_3-N content noted in surface runoff water as it flowed 2.5 km in a small drainage canal?
2. Could as much as 10 tons of dissolved salts per hectare be transferred to ground water supplies from overlying soil and tills, or vice versa, during a one and one-half year period?
3. How much salt moves from these irrigated areas to nearby rivers via [illegible] percolation?

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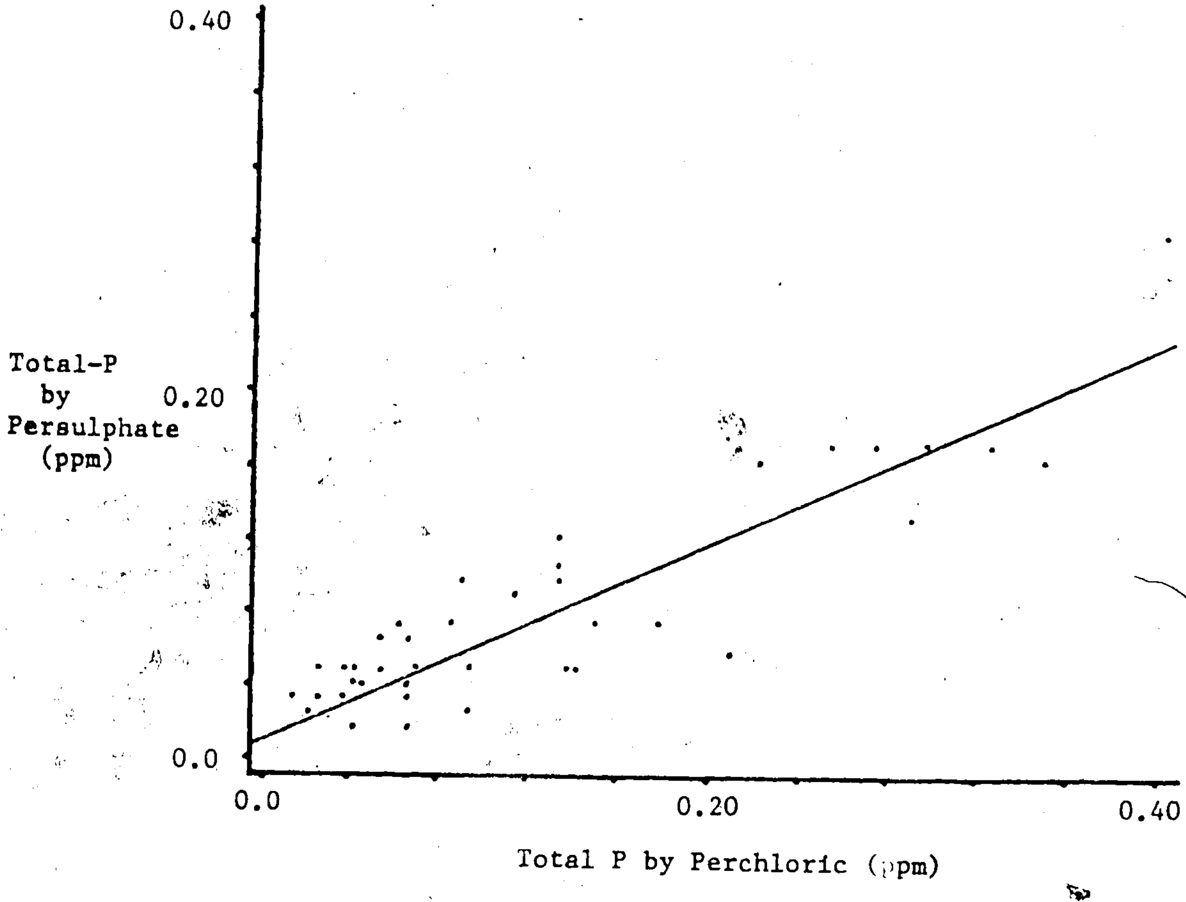
Appendix I. Some fertility data on the Hays and Vauxhall soils, means.

	1975				1976	
	Total-N (%)	NH_4-N (ppm)	NO_3-N (ppm)	Available-P (ppm)	NO_3-N (ppm)	Available-P (ppm)
Hays						
0 - 30 cm	0.11	18.7	2.16	12.0	2.55	5.94
30 - 60 cm	0.08	11.8	3.47	5.78	1.74	1.72
60 - 90 cm	0.07	10.8	3.45	3.23	2.31	1.17
90 - 180 cm	0.06	11.3	2.33	2.37	1.61	1.13
180 - 240 cm	0.05	12.3	2.07	1.66	3.08	1.95
Vauxhall						
0 - 30 cm	0.12	15.[illegible]	[illegible].83	28.6	7.33	25.4
30 - 60 cm	0.08	8.61	[illegible].36	10.9	3.13	4.36
60 - 90 cm	0.06	7.[illegible]2	1[illegible]	6.59	3.97	2.24
90 - 180 cm	0.05	7.87	[illegible]	4.96	3.64	2.38
180 - 240 cm	0.05	9.73	4.56	5.13	1.31	1.82



Statistics. . .

Correlation (r) = 0.89
Significance = 0.01
Intercept (a) = 0.021

r squared = 0.79
Std. Err. of Est. = 0.027
Slope (b) = 0.497

Appendix II. The relationship of total phosphorus by perchloric acid digestion to total phosphorus by persulfate digestion, determined on water samples from the study areas.