Water Authority
of Western Australia

WATER RESOURCES DIRECTORATE

Models to Predict the Effects of Land Disturbances on Stream Salinity in South-West Western Australia

Report No. WH 17
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ABSTRACT

A model developed by Peck (1976) to predict the effects of land disturbances on stream salinity has been re-applied to the jarrah forest region with improved parameter estimates. Validation on Wights experimental catchment suggests that the model is capable of reliable predictions in the case of agricultural clearing, although it is sensitive to variation of parameter estimates. When applied to rainfall zones of the northern jarrah forest, the model predicted that agricultural clearing would result in average stream salinity increases of ~ 100 mg L^{-1} for the high rainfall zone (> 1100 mm yr^{-1}), of ~ 400 mg L^{-1} for the intermediate rainfall zone (900 - 1100 mm yr^{-1}), and of ~ 4500 mg L^{-1} for the low rainfall zone (< 900 mm yr^{-1}). These results were sensitive to the range of conditions within each rainfall zone and to parameter estimates.

In the case of bauxite mining followed by reforestation, the model is limited to consideration of long term effects, and neglects transient effects. Predicted stream salinity increases were considerably smaller than for agricultural clearing, primarily because mining involves clearing smaller areas, and these areas are reforested.

A development of the model to include changes in throughflow and overland flow as a result of agricultural clearing resulted in predictions of salinity increases approximately 10% less than the original model.

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1. INTRODUCTION

The jarrah forest of south-west Western Australia provides the major surface water supply of the region. However the jarrah forest is under multiple land use pressures as well as natural disturbances. The major disturbances are agricultural development, bauxite mining, forest thinning and dieback disease (Phytophthora cinnamomi). In the eastern lower rainfall areas (<900 mm yr^{-1}), clearing of the forest for agricultural development has resulted in both land and stream salinisation, rendering some 34% of the divertible surface water resources of the region unpotable (>1000 mg L^{-1} total soluble salts) (Stokes and Loh, 1982). Agricultural development within water catchment areas is now restricted, and some catchments in which major water supplies have deteriorated are being reforested.

Bauxite mining is a relatively recent activity, beginning in 1963. To 1984 some 3400 ha of jarrah forest had been cleared for mining and the current rate of clearing is 300 ha yr^{-1} (Steering Committee for Research on Land Use and Water Supply, 1984). Since ore bodies tend to occur as isolated pockets on the flanks of ridges, only a portion of the forest is cleared. Unlike agricultural development, bauxite mining is followed by reforestation of the minesites. Thus bauxite mining is both a transient disturbance for the period from clearing to maturation of replanted forest, and also a permanent disturbance in removing significant depths of soil (4-5 metres typically) and introducing new species in rehabilitation. Bauxite mining also incurs the risk of spreading and intensifying dieback disease in the unmined forest, and consequently is an additional risk to stream water quality in saline areas. In the high rainfall zone (>1100 mm yr^{-1}) dieback is already very extensive (Forests Dept., 1982) and the potential for further damage is limited. However, in the eastern forest there remains considerable scope for spread but its likely impact is not yet clear (J.R. Bartle and G.C. Slessar pers. comm.).

From the above comments a need is apparent for the ability to predict the effect of major land disturbances on the salinity of water resources, particularly in the eastern more saline areas of the jarrah forest. A first attempt to do this was carried out with a simple model by Peck (1976). This model is based on the conservation of solute mass and assumes that increased salinity following clearing is derived from increased groundwater discharge to the stream at the soil solute concentration, with the increase in groundwater discharge being equal to the increase in recharge. The model neglects dynamic aspects of the problem, such as the time lag between clearing and discharge of groundwater to a stream, the effect of vegetation regrowth on rehabilitated minesites, and the progressive mining and rehabilitation sequence in a catchment. The model also neglects changes in surface runoff whose inclusion could have the effect of reducing stream salinity.

To overcome the dynamical limitations cited above, Peck _et al_. (1977) developed a more complex model. A mathematical analysis was used to estimate groundwater discharge to streams from either mined or diseased areas. Three cases of groundwater recharge were examined, namely mining and reforestation, dieback disease with no reforestation and dieback disease with reforestation. Groundwater salinities were taken from bore observations. An artificial mining sequence was adopted which is much faster than current rates. The area of dieback disease was assumed to be three times that mined. The calculation of stream salinity was based on estimates of solute and water yields from a number of zones in the catchment modelled. The model was applied to the South Dandalup Dam catchment and predicted that bauxite mining would have a negligible effect on stream salinity. However dieback disease without reforestation could result in a near doubling of stream solute concentration, but reforestation of the diseased areas would eventually return stream salinities to pre-mining values after peaking at about 1.5 times the initial mean stream salinity.

A number of model limitations were noted by the authors, including approximations in the groundwater analysis, rates of groundwater recharge, groundwater salinity values, annual variations and changes in surface runoff.

An attempt to overcome some of the limitations of the above models was made by Loh and Stokes (1980). They developed a model to predict the impact of agricultural development on annual salinity and flows into the Wellington reservoir since 1900. The basic inputs to the model were clearing history and annual rainfall since 1900. A series of relationships were developed for rainfall-runoff, surface and groundwater discharge, and surface and groundwater salinity for forested and cleared areas. Increases in surface runoff, annual variations in flow and salt load, and delays between groundwater recharge following clearing and groundwater discharge to streams were thus included. Although reasonable predictions of reservoir salinites were obtained, the authors noted a significant overprediction during the 1950's.

More recently a numerical, physically-based parametric model has been developed, known as the Darling Range Catchment Model (Hopkins, 1984; Hammond and Mauger, 1985a, b). The model predicts streamflow, groundwater flow and chloride discharge, and has the ability to take account of forest clearing, reforestation and seasonal leaf area changes. Landsat data is used to estimate leaf area index and crown cover. Chloride discharge is derived from rainfall chloride input and soil chloride mobilisation. The model has been applied to 11 experimental catchments in the northern jarrah forest ranging in area from 55 ha to 682 ha and a smaller number of large catchments ranging from 27 km^2 to 670 km^2. In each case the model is calibrated on the data record by trial and error manipulation of parameter values. Good fits to annual flows were obtained. Goodness of fit of stream salinity was only reported for one large catchment : the flow-weighted mean salinity for a 20 year calibration period was within 10% of the measured value. Following calibration, the model was used to predict the effects of a range of clearing patterns.

This report utilizes Peck's 1976 model which has the advantages of ease of prediction for a range of conditions and little input data. The model assumptions are simple and the results give some physical insight to the impact of land disturbances across the region. The model is validated on data that was not available in 1976 and consequently the predictions are of more use. Peck's model is applied to three major rainfall zones of the jarrah forest to predict average responses in streamflow salinity to both agricultural development and bauxite mining. The model is also extended to take account of increases in runoff following a land disturbance, but the extended model is only applied to agricultural clearing.

2. CONCEPTUAL MODELS

The two models treated in this report are illustrated in Fig. 1. The figure shows a lateritic soil profile and pathways of water movement typical of the jarrah forest. This simple conceptualisation considers the system to be hydrologically two-layered, the upper aquifer comprising surficial soils (often gravelly sand) and delimited at the base by a near-impermeable horizon, and the lower aquifer being deep saprolite where the permanent groundwater system is located. In high rainfall areas of the jarrah forest, the permanent groundwater is more developed and often contributes to streamflow. In low rainfall areas permanent groundwater covers a much smaller portion of the basement area and lies several to tens of metres below the stream invert (Loh _et_ _al_., 1984). However in all zones it is assumed that, given enough time following a major land disturbance, the groundwater will rise to intersect the stream. This is usually the case with agricultural clearing but may not be so with bauxite mining because of reforestation.

In the simpler model there is assumed to be negligible change in runoff. The land disturbance is assumed to increase recharge to groundwater which discharges at an equally increased rate. In the extended model additional runoff is taken into account, catering for solute leaching and streamflow dilution by throughflow and overland flow.

The models are based on the principle of conservation of mass and are fundamentally similar to the analysis of Pinder and Jones (1969). Detailed mechanisms of water and solute transport are not considered. The solutes are considered to be passive and to be transported primarily by convection. Calculations are based on yearly intervals, and lags between inflows and outflows are neglected.

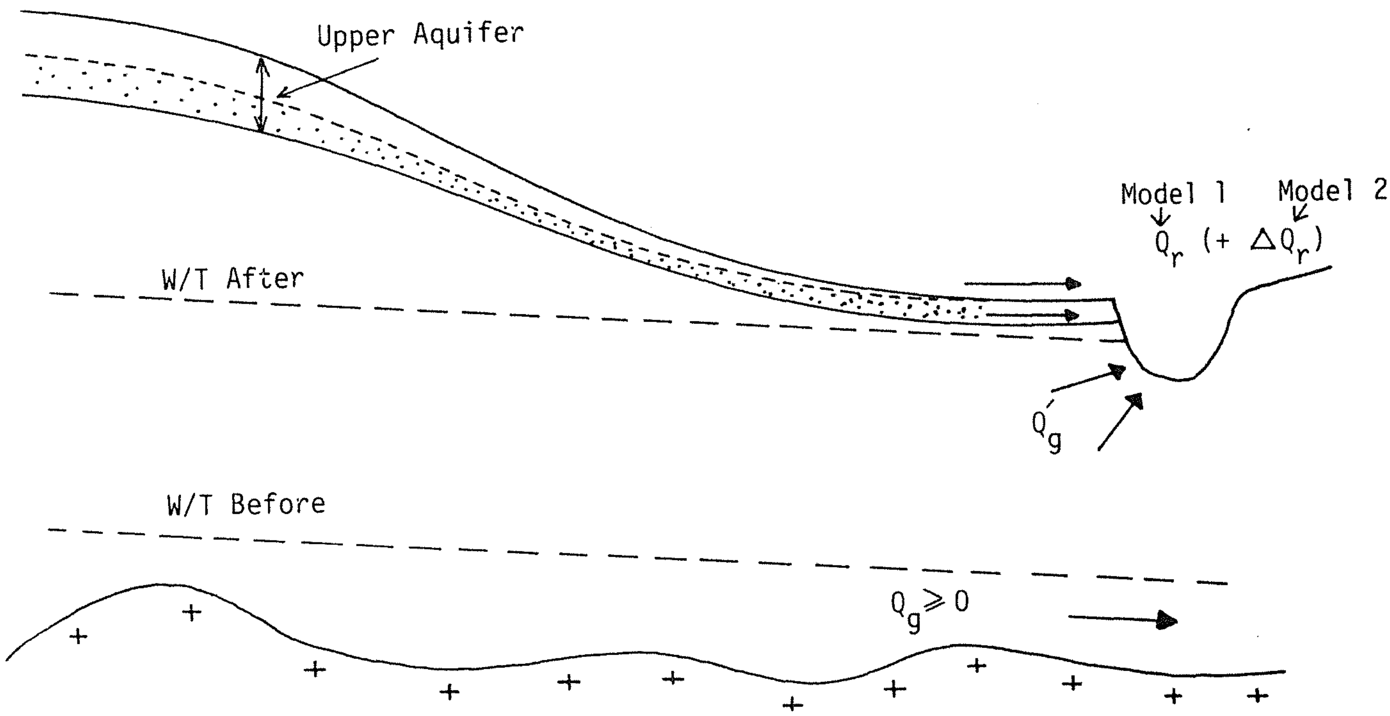


Figure 1: Schematic of Conceptual Models

3. MODEL 1

3.1 Mathematical development

Following Peck (1976), let Q_g, Q_r and Q_s represent the initial fluxes of groundwater, surface runoff and streamflow respectively. Neglecting changes of storage of water within the stream itself, we have

$$Q_g + Q_r = Q_s \tag{1}$$

If the solute concentration in groundwater, surface runoff and streamflow are c_g, c_r and c_s respectively, then

$$c_g Q_g + c_r Q_r = c_s Q_s \tag{2}$$

Let a change of land use alter the water fluxes by ΔQ_g, ΔQ_r and ΔQ_s. Then

$$Q_g + \Delta Q_g + Q_r + \Delta Q_r = Q_s + \Delta Q_s \tag{3}$$

If the solute concentrations of the additional groundwater and surface runoff are c_g' and c_r', and c_s' is the solute concentration in the stream, then

$$c_g Q_g + c_g' \Delta Q_g + c_r Q_r + c_r' \Delta Q_r = c_s'(Q_s + \Delta Q_s) \tag{4}$$

Subtracting equation 2 from equation 4 gives

$$c_g' \Delta Q_g + c_r' \Delta Q_r = c_s' Q_s + c_s' \Delta Q_s - c_s Q_s$$

Adding $- c_s \Delta Q_s$ to both sides

$$c_g' \Delta Q_g + c_r' \Delta Q_r - c_s \Delta Q_s = c_s' Q_s + c_s' \Delta Q_s - c_s Q_s - c_s \Delta Q_s$$

$$= (c_s' - c_s)(Q_s + \Delta Q_s).$$

Rearranging and substituting for ΔQ_s gives

$$\Delta c_s = c_s' - c_s = \frac{c_g'\Delta Q_g + c_r'\Delta Q_r - c_s\Delta Q_g - c_s\Delta Q_r}{Q_s + \Delta Q_g + \Delta Q_r}$$

$$\text{or} \quad \Delta c_s = \frac{(c_g'-c_s)\Delta Q_g - (c_s - c_r')\Delta Q_r}{Q_s + \Delta Q_g + \Delta Q_r} \tag{5}$$

Because of the difficulty in estimating ΔQ_r and c_r', it is assumed that ΔQ_r is negligible, and equation 5 becomes

$$\Delta c_s = \frac{(c_g' - c_s)\ \Delta Q_g}{Q_s + \Delta Q_g} \tag{6}$$

The assumption that ΔQ_r is negligible would be valid if

$$|\Delta Q_r| < |\Delta Q_g\ (c_g'-c_s)/(c_s-c_r')| \tag{7}$$

$$\text{and} \quad |\Delta Q_r| << |\ Q_s + Q_g| \tag{8}$$

Conditions 7 and 8 are generally not met as is evidenced from the results of clearing on Wights catchment (Stokes and Loh, 1982) and other cleared or mined catchments (Loh _et al_., 1984). Thus equation 6 will tend to overestimate Δc_s.

In equation 6, c_s and Q_s are determined from stream gauging and sampling prior to the land use change. c_g' can be determined from baseflow measurement or soil solute concentration. ΔQ_g is estimated by

$$\Delta Q_g = A_c\ \Delta G \tag{9}$$

where A_c is the area over which the land use change has taken place and ΔG is the change in groundwater recharge associated with the land use change. ΔG is determined by field measurement.

Substituting equation 9 into equation 6 gives

$$\Delta c_s = A_c \, \Delta G \, (c_g' - c_s)/(Q_s + A_c \, \Delta G) \tag{10}$$

For ease of computation of equation 10, some additional parameters are defined:

$$\alpha = A_c/A, \ \beta = \Delta G/P, \ \gamma = Q_s/AP$$

where P is precipitation.

Substituting these parameters into equation 10 gives

$$\Delta c_s = \alpha\beta(c_g' - c_s)/(\gamma + \alpha\beta)$$

$$\text{or} \quad \Delta c_s = (c_g' - c_s)/(\gamma/\alpha\beta + 1) \tag{11}$$

Defining $\varepsilon = (\gamma/\alpha\beta + 1)^{-1}$ gives

$$\Delta c_s = \varepsilon(c_g' - c_s) \tag{12}$$

Two limits to equations 11 and 12 are apparent:

1. $\Delta c_s \rightarrow 0$ as $c_g' \rightarrow c_s$
2. $\Delta c_s \rightarrow c_g' - c_s$ as $\varepsilon \rightarrow 1$ or $\gamma/\alpha\beta \rightarrow 0$

Limit 1 is found to be approached in the high rainfall zone while limit 2 is approached in the low rainfall zone.

3.2 Model Validation

To investigate the predictive accuracy of the model, it is applied to two catchments which have undergone land use changes in the form of clearing for agricultural development and bauxite mining followed by rehabilitation. The catchments involved are Wights and More Seldom Seen. Of these Wights is the most significant test because of superior data.

Wights catchment

Wights catchment is 93.8 ha in area and lies within 1 kilometre of Wellington Reservoir (115° 58' E, 33° 24' S) in south-western Australia. It was established in 1974 as one of a pair of catchments naturally vegetated by eucalypt forest. In the summer of 1976/77 Wights was totally cleared for agricultural development. The merchantable timber was removed first and the remaining timber heaped and burnt. In May-June 1977 the catchment was aerially sown to clover and grass. Since 1978 the catchment has been grazed with sheep and horses. Further agricultural developments involving tree stump removal, pinwheel raking, bulldozer levelling and ploughing were carried out in the autumn of 1980. The paired catchment, Salmon, has remained as undisturbed forest.

For both catchments, continuous streamflow measurement and daily water quality sampling has continued since May 1974. On Wights catchment 35 deep piezometers were installed to monitor groundwater dynamics and groundwater quality.

The model parameter values for Wights catchment are given in Table 1 and discussed below.

Table 1 Model parameter values for Wights catchment

parameter	value
α	1
β	0.059
γ	0.102
c_g'	988 mg L^{-1}
c_s	201 mg L^{-1}

α requires no discussion since the catchment was totally cleared. The value of the change in recharge (ΔG) is taken from Peck (1983) as 60 mm. This is the mean estimated increase in recharge for the period 1977-81. The calculation is based on catchment average groundwater level changes and a storage coefficient derived by Bestow (1975) at Del Park. Since a storage coefficient has not been determined at Wights catchment, the recharge values may be in significant error. β is determined by dividing ΔG by the mean rainfall for the period (1022 mm).

The ratio of mean annual streamflow to rainfall is determined from the 3 calibration years on Wights (1974-77) and then adjusted to the 10 year trend on the undisturbed Salmon catchment. For the period 1974-77 on Salmon, γ is 1.25 times greater than the 10 year value, which is taken to be close to the long term mean because mean annual rainfall over 1974-84 is very close to the long term mean rainfall (97%). The 1974-77 γ on Wights is therefore reduced by 20%. To determine the solute concentration of the additional groundwater discharge, summer baseflow concentrations are used. There is a tendency for the baseflow concentration to increase during the summer, probably due to evaporative concentration. The baseflow solute concentration was calculated as an average from December to February for the period 1977-82, as shown in Table 2. The mean baseflow chloride concentration for the period is converted to TSS by the following equations (R. Stokes pers. comm.).

$$\mathrm{TSS} = 1.66\ \mathrm{Cl}^- + 48.3 \qquad \mathrm{TSS} > 340\ \mathrm{mg\ L}^{-1} \tag{13}$$
$$\mathrm{TSS} = 1.9\ \mathrm{Cl}^- + 9.9 \qquad \mathrm{TSS} < 340\ \mathrm{mg\ L}^{-1} \tag{14}$$

Table 2 Monthly mean baseflow Cl^- concentrations (mg L^{-1}) for Wights catchment

	Dec	Jan	Feb	Mean
1977/78	539 (11)*	596 (10)	712 (10)	616
1978/79	411 (11)	519 (6)	634 (10)	521
1979/80	561 (12)	638 (9)	698 (9)	632
1980/81	498 (5)	585 (4)	624 (14)	569
1981/82	455 (13)	531 (9)	-	493
			period mean	566

* number in brackets is number of samples

The mean streamflow solute concentration prior to land distrubance (c_s) is taken as the flow-weighted mean of Wights for the period 1974-76. This value requires no adjustment to the long term mean of Salmon catchment because Salmon's 1974-76 value is identical to the 1974-84 value.

To determine the actual change in streamflow solute concentration following the land change on Wights, 3-year flow-weighted moving average concentrations are calculated and plotted in Fig. 2. Subtracting the mean of the last four points from the single pre-clearing values, gives an increase in streamflow TSS of 228 mg L^{-1}. Substituting the parameter values in Table 1 into equation 11 gives a predicted change of Δc_s = 288 mg L^{-1}. Considering the level of uncertainty in the parameter values, the prediction is remarkably close to the observed value. The use of this model is expected to predict higher than observed concentration changes because changes in overland flow and throughflow are ignored, and these would generally have the effect of reducing the solute concentration. Of more importance than the single predicted value is the sensitivity of the results to parameter estimates. To establish this a sensitivity analysis is conducted.

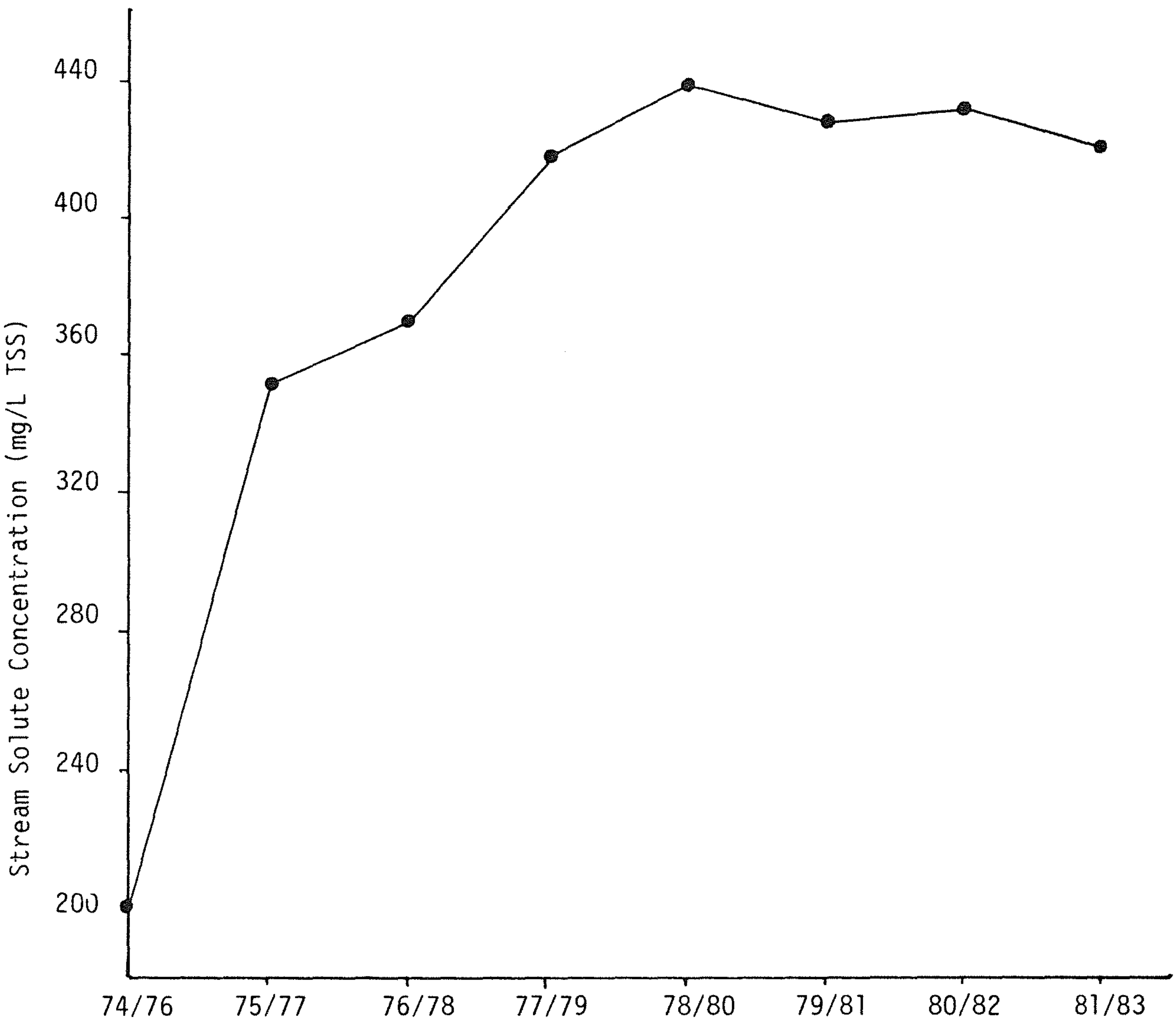


Figure 2: Three-Year Moving Average Stream Solute Concentration (TSS) Observed at Wights Catchment

Sensitivity analysis

In the analysis the sensitivity of each parameter is tested individually first and then jointly to estimate maximum prediction errors. The range of the parameter values to be tested and the model prediction results are shown in Table 3. Values of parameters not being tested are as in Table 1.

Table 3 Sensitivity analysis of individual parameters

parameter	range	predicted Δc_s (mg L^{-1})
β	.088-.029	365-174
γ	.2-.05	179-426
c_g'	1300-700	403-183
c_s	300-100	252-325
α	.5-.1	177-43

It is evident from Table 3 that the parameters β, γ and c_g' are approximately equally sensitive, given that the assumed parameter value ranges are realistic. The value of c_s is less critical. Two values of α were selected to test the effect of part clearing. 50% clearing resulted in a 40% reduction in the increase of stream salinity, whilst 10% clearing reduced the increase by 85%.

The maximum prediction errors may be estimated by selecting parameter values from Table 3 which maximise or minimise the prediction:

	β	γ	c_g'	c_s	α	predicted Δc_s
maximum prediction :	.088	.05	1300	100	1	765 mg L^{-1}
minimum prediction :	.029	.2	700	300	1	51 mg L^{-1}

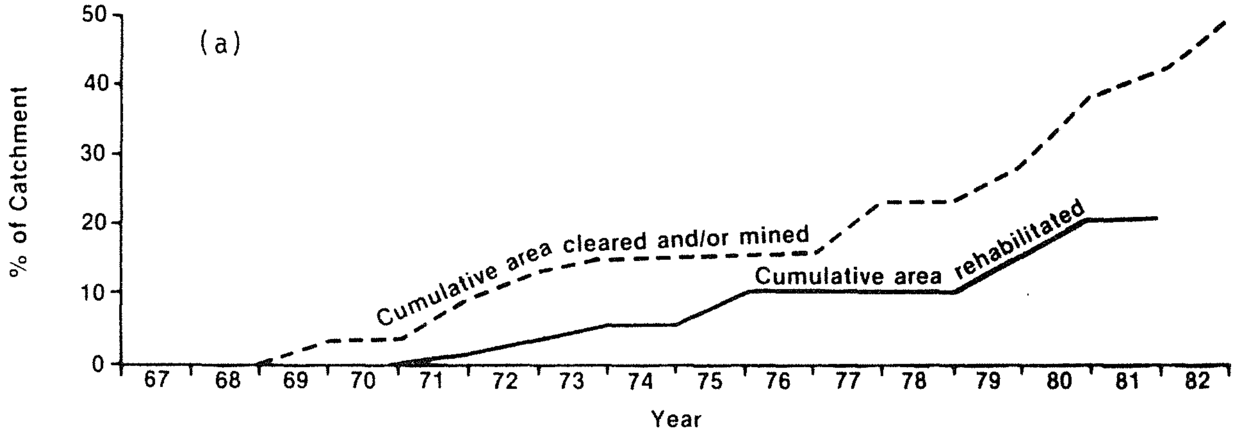
More Seldom Seen Catchment

More Seldom Seen Catchment is one of three catchments established in 1966 that have been used to evaluate the effects of bauxite mining and rehabilitation on the surface hydrology of the high rainfall zone. Clearing began in More Seldom Seen catchment in 1969 and continued over the following 13 years to reach nearly 50% of the catchment (Fig. 3). Rehabilitation began in 1971 and will continue until all the mined and cleared areas are rehabilitated (Fig. 3). Clearing, mining and rehabilitation has resulted in a significant increase in the streamflow yield of More Seldom Seen catchment relative to Waterfall Gully (Loh et al., 1984). Comparing the pre-1973/74 water years to the following 8 years indicates an average streamflow yield increase of 40%. However, virtually no detectable trend has taken place in the streamflow solute concentration as shown in Fig. 3. Comparing pre- and post-1973/74 water years as previously shows a slight decrease of 7 mg L^{-1} TSS in mean concentration.

The parameter values for the model prediction and their method of determination are given in Table 4.

Table 4 Parameter values for More Seldom Seen catchment

parameter	value	method of determination
α	0.5	Loh et al. 1984
β	0.061	equation 15 with P = 1200 mm
γ	0.216	Loh et al. 1984
c_s	134 mg L^{-1}	mean value 1966-73, Loh et al. 1984
c_g'	180 mg L^{-1}	Peck 1976



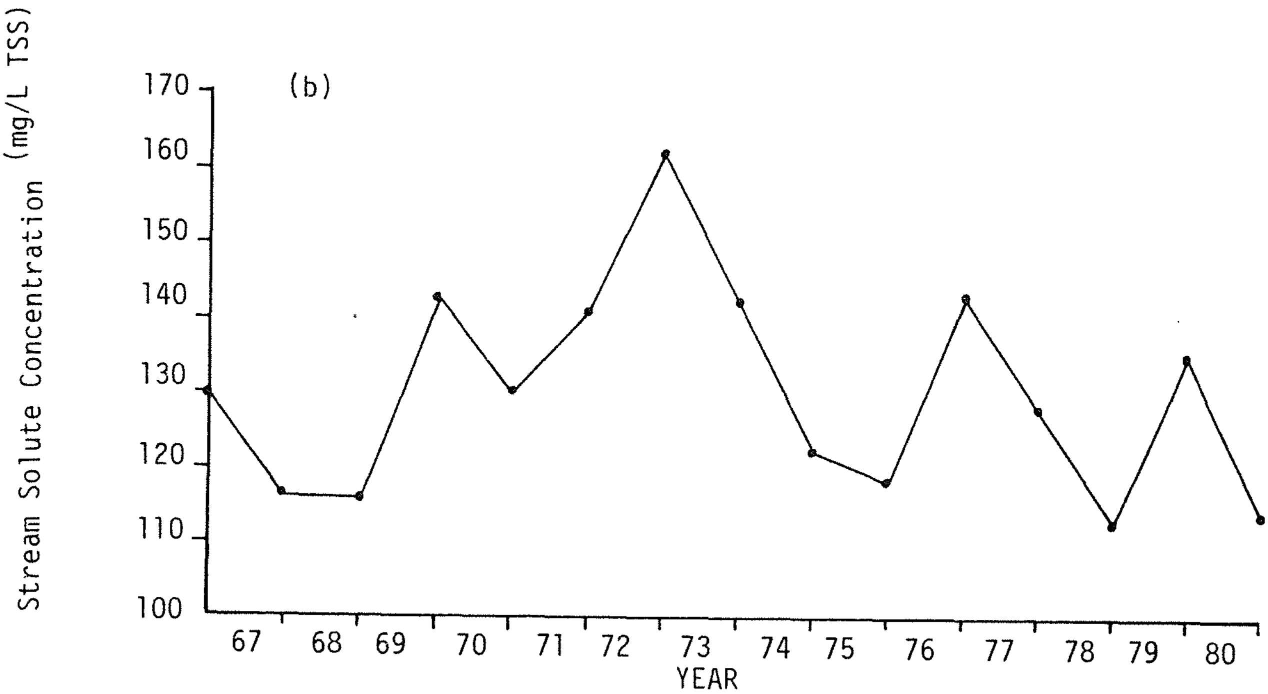


Figure 3: Areas Mined and Rehabilitated (a) and Observed Stream Solute Concentration for More Seldom Seen Catchment (b)

The model prediction for the above parameters is

$$\Delta c_s = 6.7 \text{ mg L}^{-1}$$

This is a maximum prediction in the sense that no forest rehabilitation is taken into account in β. Nevertheless the model does indicate that there would be virtually no detectable change for this catchment.

3.3 Application of the model to agricultural clearing and bauxite mining in the jarrah forest

One of the main uses of the model is to explore the potential effects of major land disturbances on different parts of the jarrah forest. Two land disturbances are considered here, bauxite mining and agricultural clearing. Bauxite mining is not a permanent land use change because it is followed by forest rehabilitation. Both of these changes may lead to a permanent increase in groundwater recharge. Clearing for agricultural development is, of course, a permanent land use change.

For management purposes the jarrah forest is often divided into three zones based on rainfall, as shown below:

high rainfall zone (HRZ)	>1100 mm yr^{-1}
intermediate rainfall zone (IRZ)	1100 - 900 mm yr^{-1}
low rainfall zone (LRZ)	<900 mm yr^{-1}

Application of the model requires parameter estimates for each zone for both bauxite mining and agricultural clearing. The derivation of parameter estimates are discussed below:

α - for bauxite mining α is assumed to be 0.3 in the HRZ, 0.2 in the IRZ and 0.05 in the LRZ (G. Slessar pers. comm.). However, within each rainfall zone α is known to vary considerably from site to site. For agricultural clearing α is taken to be 1, although in some circumstances it may be significantly less.

ß - two measurements of change in recharge (ΔG) following clearing have been made by Peck (1983) in different rainfall zones:

	mean annual rainfall for period	change in annual recharge (ΔG)	ß
Wights	1022 mm	60 mm	0.059
Lemon	659.5 mm	30 mm	0.045

These two measurements are used to form a linear extrapolation equation for the region, which takes the form

$$\Delta G = 0.0828 P - 24.6 \tag{15}$$

Equation 15 is used to calculate ß for a given mean annual rainfall.

For bauxite mining two estimates of ß may be derived from the analyses of Briese (1979) and Peck (1983). Briese calculated infiltration values in a mined and rehabilitated area, and a control forested area, at the Del Park mine site. On the mined area he showed that a large increase in infiltration occurred immediately after mining but this decreased to somewhat smaller values within 3-4 years (Table 5). Assuming that 1976-77 and 1977-78 represent 'long-term' changes in groundwater recharge, an average ß value of 0.015 is obtained. Naturally a more extensive period of data collection would be necessary to determine the long term effect of rehabilitation. The data set was also affected by the significantly lower than average (approx. 1300 mm yr^{-1}) annual rainfall from 1975-78.

Table 5 Estimated changes in groundwater recharge following mining* at Del Park (adapted from Briese, 1979)

	1974-75	1975-76	1976-77	1977-78
average infiltn. in forest bores (mm)	124.5	94.5	88.8	97.9
rainfall (mm)	1591	1060	788	873
ΔG (mm)	97.3	46.6	8.5	16.1
ß	0.061	0.044	0.011	0.018
			mean = 0.015	

* mining and rehabilitation took place in 1972-73

Peck (1983) analysed bore hydrographs at the Del Park research catchment (approx. 3 km from the above site) over the period 1975-81. 14 of the bores were located in an area mined and rehabilitated mostly in 1977, and a control group of 20 bores was located some 400 m away in forest. Over the period 1975-81 the mean water level of the mine bores rose 2.05 m relative to the mean water level of forest bores. Assuming a storage coefficient of 0.039 (Bestow, 1975), this corresponds to a mean annual groundwater recharge increase of 13.3 mm yr^{-1} and a ß of 0.011. However Peck points out that, at this site, lateral flow in the groundwater aquifer is significant and consequently groundwater recharges derived in the above way could be unreliable. Nevertheless the ß value is of the same order of magnitude as that derived from Briese's data. An average of the two values is assumed here (ß = 0.013).

γ - to calculate γ for each rainfall zone, values of long term mean annual rainfall and streamflow are required for catchments lying wholly in one particular zone. Such catchments and the associated values are given in Table 6 (Loh pers. comm.). The mean and range of γ values for each zone are also presented in Table 7.

Table 6 Catchment long term mean annual rainfall and streamflow

catchment	area (km^2)	period of record (yrs)	rainfall (mm)	streamflow (mm)	zone	γ
Hansen	0.85	4	1340	60	HRZ	0.045
Higgins	0.71	4	1300	70	HRZ	0.054
Lewis	1.70	4	1350	90	HRZ	0.067
Warren	0.84	4	1325	160	HRZ	0.121
Bennett	0.85	4	1330	230	HRZ	0.173
Stones	14.7	9	1210	120	HRZ	0.099
Falcon	5.46	7	1200	110	HRZ	0.092
Carey	2.70	5	1425	370	HRZ	0.260
Lewin South	1.05	5	1230	180	HRZ	0.146
Salmon	0.082	7	1190	180	HRZ	0.151
Lewin North	1.23	5	1230	240	HRZ	0.195
Clarke	17.1	19	1200	299	HRZ	0.249
Falls	28.7	7	1220	340	HRZ	0.279
Logue	34.4	23	1250	405	HRZ	0.324
April South	2.08	5	1080	110	IRZ	0.102
April North	2.36	5	1060	100	IRZ	0.094
March	2.73	5	1050	120	IRZ	0.114
Ludlow	10.1	4	980	50	IRZ	0.051
Pickering	31.1	11	1000	70	IRZ	0.070
Margaret	14.84	4	960	85	IRZ	0.089
St Pauls	24.8	6	940	90	IRZ	0.096
Yerramminup N	2.42	6	840	40	LRZ	0.048
Yerramminup S	1.96	6	840	40	LRZ	0.048
Tunnel	2.07	6	740	0	LRZ	0.0
Bee Farm	1.81	6	740	0	LRZ	0.0
Ernies	2.70	7	725	1	LRZ	0.0014
Wellbucket	4.65	7	720	0	LRZ	0.0
Yarra Rd	6.28	7	720	1	LRZ	0.0014
Encampment	0.16	11	610	10	LRZ	0.016

Table 7 Mean and range of γ values for each rainfall zone

zone	mean	range	number
HRZ	0.161	0.324 - 0.045	14
IRZ	0.088	0.114 - 0.051	7
LRZ	0.014	0.048 - 0.0	8

c_g' - the groundwater salinity after clearing is estimated from soil solute concentrations of deep bores in the particular rainfall zones. Stokes et al. (1980) showed that at cleared sites the groundwater salinities approach the soil solute concentration, particularly when rainfall is less than 1100 mm yr^{-1}. These authors also summarised soil solute concentrations at 31 sites throughout the Darling Range. Slessar et al. (1983) reported Alcoa's soil salt storage data in the northern jarrah forest. These two sources of data have been combined in Table 8 to determine solute concentration statistics for each rainfall zone.

Table 8 Summary statistics for soil solute concentration TSS (mg L^{-1})

zone	mean	range	number
HRZ	581	140-2226	42
IRZ	1374	350-2176	12
LRZ	6092	2332-14777	18

c_s - the mean streamflow salinities of fully forested catchments by rainfall zone have been presented by Loh et al. (1983) and are repeated in Table 9, along with ranges for each zone.

Table 9 Mean and ranges of streamflow salinities by rainfall zone

zone	mean salinity ($mg\ L^{-1}$ TSS)	range of salinity ($mg\ L^{-1}$ TSS)	number of catchments
HRZ	129	90 - 190	16
IRZ	192	110 - 260	8
LRZ	107	80 - 130	6

A summary of the parameter values for each rainfall zone for bauxite mining and agricultural clearing is given in Table 10.

Table 10 Parameter values for model application to rainfall zones

parameter		HRZ			IRZ			LRZ		
		max	min	mean	max	min	mean	max	min	mean
α	mining	.5	.1	.3	.4	.1	.2	.2	.01	.05
α	agriculture*	1	.5	.8	1	.5	.8	1	.5	.8
β	mining *	.03	.0	.013	.03	.0	.013	.03	.0	.013
β†	agriculture	.065	.060	.062	.062	.055	.058	.055	.048	.052
γ	both	.324	.045	.161	.114	.051	.088	.048	.0	.014
c_g'	both	2226	140	581	2176	350	1374	14777	2332	6092
c_s	both	190	90	129	260	110	192	130	80	107

* ranges assumed

† ranges determined from variation in P, not ΔG

Results

agricultural clearing

Model predictions of the increase in stream salinity for each rainfall zone may be obtained using the parameter values of Table 10. The 'average' response of each rainfal zone was calculated using the mean values in Table 10 which gave the following results:

$$\begin{aligned} \text{HRZ} &: \Delta c_s = 106 \text{ mg L}^{-1} \\ \text{IRZ} &: \Delta c_s = 408 \text{ mg L}^{-1} \\ \text{LRZ} &: \Delta c_s = 4478 \text{ mg L}^{-1} \end{aligned}$$

As shown in Table 10, there is considerable variability in parameter values in any given rainfall zone. The effect of this variability on model prediction was tested for each parameter individually (with the other parameters taking mean values). The results are shown in Table 11. The groundwater salinity was seen to be the most sensitive parameter and the streamflow percentage (γ) the next. The proportion of area cleared (α) had moderate effect on the predictions. The variation of β had little effect, but this is probably because only the range in P and not ΔG was included. The mean streamflow salinity also had little effect on the predictions. To obtain predictions for the most extreme situations, the parameter values producing maximum and minimum responses were used simultaneously, and the results are shown in the bottom row of Table 11.

Table 11 The variation in stream salinity increases predicted within each rainfall zone as a result of agricultural clearing

parameter	HRZ max	HRZ min	IRZ max	IRZ min	LRZ max	LRZ min
α	126	73	470	293	4715	3890
β	110	104	426	394	4540	4386
γ	237	60	563	342	5985	2779
c_g'	494	3	685	55	10976	1665
c_s	116	92	436	385	4478	4461
all parameters	1262	-4	1134	17	14697	734

To summarise the results for agricultural clearing, moderate increases in stream salinity are predicted for the HRZ and IRZ, and very large increases for the LRZ. However there may be significant variability of response within each rainfall zone, depending on the specific attributes of a catchment.

bauxite mining

Predictions of increases in stream salinity can also be made for the long term effects of bauxite mining and reforestation. The 'average' responses of each rainfall zone are given below, and the variations within each zone are given in Table 12.

HRZ : $\Delta c_s = 11 \text{ mg L}^{-1}$

IRZ : $\Delta c_s = 34 \text{ mg L}^{-1}$

LRZ : $\Delta c_s = 266 \text{ mg L}^{-1}$

Table 12 The variation in stream salinity increases predicted within each rainfall zone as a result of bauxite mining

parameter	HRZ max	HRZ min	IRZ max	IRZ min	LRZ max	LRZ min
α	18	4	66	17	937	55
β	24	0	75	0	579	0
γ	36	5	57	26	5985	80
c_g'	50	0	57	5	651	99
c_s	12	9	36	32	267	265
all parameters	534	0	378	0	14697	0

Comparing the above results to those of agricultral clearing, it is clear that long term stream salinity increases under bauxite mining and rehabilitation would be much less. Small increases are predicted for the HRZ and IRZ and moderate to large increases for the LRZ. If reforestation reduces increases in recharge to zero, then no long term salinity increases would be expected. This is the assumption made by Peck et al. (1977) who predicted negligible effects from bauxite mining and reforestation alone.

dieback disease

The fungus Phytophthora cinnamomi is a soil/water borne root invading pathogen that causes the disease jarrah dieback. On susceptible sites the disease results in complete mortality of the overstorey and dramatic decline in the diversity and vigour of the understorey (Podger, 1972). Dieback now occurs over some 14% of the forest, mostly in the HRZ (Forests Dept., 1982). Severely affected sites with extensive jarrah overstorey mortality probably make up about half of the infected area.

Bauxite mining clearly has the potential to spread dieback disease, although as mining has been restricted to five locations covering less than 2% of the northern jarrah forest, its contribution thus far to the total dieback impact has been very small. In its report to the Environmental Protection Authority on the Wagerup alumina project, the Technical Advisory Group (1978) estimated that mining in the HRZ near Wagerup could result in the spread of up to one hectare of dieback for each hectare mined. More recent estimates, although still qualitative, suggest the actual figure is substantially lower (G.C. Slessar, pers. comm.). The effect of the potential expansion of bauxite mining into the IRZ on dieback spread and intensification is not yet clear. Until further data is available, it seems reasonable to assume that a worst case situation could be that 20% of the IRZ (in the area of mining) could be lost to dieback disease. This magnitude is not likely to be attained because of the intensive dieback management procedures being developed for mining operations, and also the fact that severely degraded areas could be reforested. Comparisons of the impacts of bauxite mining alone and bauxite mining with associated 20% dieback loss on stream salinity increases (mg L^{-1}) are given below:

	HRZ	IRZ	LRZ
bauxite mining	11	34	266
bauxite mining & dieback	18	66	1128

4. MODEL 2

One of the main simplifying assumptions in Model 1 is the neglect of changes in streamflow components other than groundwater. However the examples of More Seldom Seen catchment and Wights catchment show that the major land disturbances can result in significant increases in streamflow, in these cases being respectively 40% and 120%. Such streamflow increases could significantly change the stream solute concentration.

Model 2 attempts to incorporate the effects of increased stream runoff. In this case equation 5 is appropriate:

$$\Delta c_s = \frac{(c_g' - c_s)\Delta Q_g - (c_s - c_r')\,\Delta Q_r}{Q_s + \Delta Q_g + \Delta Q_r} \qquad (5)$$

Two additional variables appear, namely ΔQ_r and c_r'.

For long periods of time (several years) it is assumed that the following relationships could be defined:

$$\Delta Q_r = \eta Q_s \qquad (16)$$

$$c_r' = \phi c_g \qquad (17)$$

Equation 17 implies that c_r' is directly related to c_g. In practice increased stream runoff is a mixture of overland flow, throughflow and groundwater flow. These sources interact with each other in the surface aquifer. During summer salt accumulates in the surface aquifer and is subsequently leached to the stream during winter. Hence the additional streamflow may be characterised by high solute concentrations in early winter and lower concentrations later. The mean annual c_r' will depend on the mechanisms and rates of salt accumulation in, and leaching from, the surface aquifer for each individual catchment. Whether or not c_r' is simply related to c_g is not known. Equation 17 is used here because c_g is a readily determinable quantity.

Substitution of equations 16 and 17 into equation 5 gives

$$\Delta c_s = \frac{c_g'\,(\alpha\beta + \phi\eta\gamma) - c_s\,(\alpha\beta + \eta\gamma)}{\alpha\beta + \gamma\,(1 + \eta)} \qquad (18)$$

where α, β and γ are defined as previously.

The best estimates of η and ϕ are obtainable from Wights catchment. Here the streamflow increase following clearing was 120%, giving $\eta = 1.2$. ϕ is more difficult to evaluate and requires an estimate of c_r. On Wights catchment seepage areas were identified and solute concentrations measured throughout one winter (Stokes and Loh, 1982). Following clearing an extensive saturated area of at least 20 ha developed during winter. Located within this area was a saline seepage with Cl^- concentrations up to 422 mg L^{-1}. Surrounding this saline seepage was a more variable and fresher seepage characterized by Cl^- concentrations of 20-56 mg L^{-1}. c_r was not measured on Wights but the value of 35 mg L^{-1} Cl^- (76 mg L^{-1} TSS) used by Stokes and Loh (1982) for c_r' would be a reasonable estimate, given the above range of values in the fresher outer seepage on Wights catchment. An assumption of no increase in c_r yields a lower bound to ϕ of 0.077. An upper estimate of ϕ may be obtained by assuming that the fresher seepage water passes through the saline seepage before reaching the stream. c_r' will then be the mean solute concentration in the saline seepage area which in this case was 373 mg L^{-1} TSS, yielding $\phi = 0.38$.

Applying equation 18 (Model 2) to Wights catchment with the parameter values of Table 1 and with $\eta = 1.2$ and $\phi = 0.38$ and 0.077 gives the following predictions:

$$\Delta c_s = 238 \text{ mg L}^{-1} \quad (\phi = 0.38)$$
$$\Delta c_s = 110 \text{ mg L}^{-1} \quad (\phi = 0.077)$$

The prediction using $\phi = 0.38$ gives a result very close to the observed change in stream salinity (228 mg L^{-1}). Using $\phi = 0.077$ gives a significantly lower value. This shows that Model 2 is sensitive to the value of ϕ and also suggests that the upper estimate is more appropriate in this case. However too much weight should not be given to this due to the uncertainties in the other parameter values.

Application of Model 2 to More Seldom Seen catchment, with the parameter values of Table 4 and with η = 0.4 and ϕ = 0.38 and 0.077 gives:

$$\Delta c_s = -13 \text{ mg L}^{-1} \quad (\phi = 0.38)$$
$$\Delta c_s = -27 \text{ mg L}^{-1} \quad (\phi = 0.077)$$

The observed change of approximately -7 mg L^{-1} again suggests that the higher value of ϕ may be more appropriate.

Application of Model 2 to the rainfall zones of the jarrah forest with the mean parameter values of Table 10 and ϕ = 0.38, η = 1.2 gives the following 'average response' salinity increase (mg L^{-1}) predictions for agricultural clearing:

	HRZ	IRZ	LRZ
Model 1	106	408	4478
Model 2	99	374	3951

With the assumed parameter values Model 2 only gives a slight reduction in predicted stream salinity increases.

5. DISCUSSION

Model 1 is identical to that of Peck (1976) but since that time more data has become available on which to validate the model. The best validation data set is Wights catchment. Comparing the parameter values of Table 1 with those of Peck (1976, Table 1), shows that Peck originally significantly overestimated the values of increased groundwater recharge and groundwater salinity. This resulted in a considerable over-prediction of the stream salinity change on Wights. However, the more reasonable values used here result in a prediction only slightly in excess of that observed. Application of Model 1 to More Seldom Seen catchment gives an identical prediction to Peck (1976), but in this case the excessive value of increased recharge used by Peck is offset by the much smaller value he used for the area cleared. Both predictions are similar to the observed value. The accurate predictions of Model 1 for the above two catchments indicates that it is an appropriate model for application to a range of conditions across the jarrah forest.

Model 2 is an extension of Model 1 that includes the effects of additional sources of increased streamflow besides groundwater. The main difficulty is assigning an appropriate value of solute concentration to these additional sources. Application of the model to the rainfall zones for agricultural clearing reduces the predicted salinity increases, but not greatly (<12%).

<u>model limitations</u>

<u>structure</u>

The models neglect dynamic aspects of the problem. This does not reflect on the accuracy of the model but the way in which it can be used. For example transient aspects of mining and reforestation cannot be modelled. Also annual variations in flows and salinities cannot be treated.

The model assumes that the land disturbance always results in groundwater tables rising and groundwater eventually contributing to streams. Although this is frequently the case for agricultural clearing, it would not necessarily be so for bauxite mining and reforestation.

<u>data</u>

The change in groundwater recharge following clearing has only been measured at two sites, and in the calculations the same storage coefficient was used, which itself was measured at a third site.

Values of mean streamflows prior to disturbance have been calculated on the basis of a few years of record.

Groundwater salinity is either measured from summer baseflow concentrations if groundwaters are known to be contributing or otherwise from mean soil solute concentrations. The first method has problems associated with evaporative concentration of groundwater flux. The latter method requires a large number of profile samples per catchment to determine mean solute concentrations, and it is not totally clear how this number is related to the salinity of groundwater discharge.

6. CONCLUSIONS

The model developed by Peck (1976) has been found, when validated on more recent data, to provide reasonable prediction accuracy for changes to stream salinity following agricultural clearing. The model is, however, sensitive to variation of parameter values.

When applied to agricultural clearing in different rainfall zones, the model predicts average salinity increases of ~100 mg L^{-1} in the HRZ, ~ 400 mg L^{-1} in the IRZ and ~ 4500 mg L^{-1} in the LRZ. Within each rainfall zone a significant range of stream salinity changes are predicted.

Application of the model to bauxite mining is limited by the exclusion of transient effects. Prediction of the long term effects of bauxite mining and reforestation indicate salinity increases of ~ 10 mg L^{-1} in the HRZ, ~ 35 mg L^{-1} in the IRZ and ~ 270 mg L^{-1} in the LRZ. Again the results are sensitive to parameter estimates and to the range of conditions existing within each rainfall zone. Insufficient data is currently available to realistically predict the effects of the spread and/or intensification of dieback disease associated with mining.

The main model limitations are the neglect of dynamic aspects of the problem and the assumption that land disturbance always results in groundwater contributing to streams. Data limitations affect estimates of increased groundwater recharge, groundwater salinity, and to a lesser extent mean streamflows and stream salinities prior to disturbance.

Model 2 extends Peck's 1976 model to include additional sources of streamflow increase following land disturbance. Predictions by this model for agricultural clearing indicate a reduction of Model 1 predictions of stream salinity increases by about 10%.

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