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WATER YIELD RESPONSE TO LAND USE CHANGE IN SOUTH-WEST WESTERN AUSTRALIA

DEPARTMENT OF ENVIRONMENT
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SALINITY AND LAND USE IMPACTS

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WATER YIELD RESPONSE TO LAND USE CHANGE IN SOUTH-WEST WESTERN AUSTRALIA

by

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Resource Science Division

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SALINITY AND LAND USE IMPACTS SERIES

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Cover photograph:
Harvey River upstream of Stirling Dam by
Mark Pearcey

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Summary

The forested areas of south-west Western Australia produce little streamflow from moderate rainfall. The water yield from forested experimental catchments averages 104 mm (10.6% of annual rainfall) while from the surface water supply catchments for Perth it averages 71 mm (7% of annual rainfall). The low runoff is attributed to the large soil water storage available for continuous use by the forest vegetation.

Studies in 27 experimental catchments in the south–west of Western Australia examined the impacts of land use practices such as clearing for agricultural development, forest harvesting and regeneration, forest thinning, bauxite mining and reforestation on water yield.

A permanent reduction in vegetation cover by clearing for agriculture has led to permanent increases of water yield of about 30% of annual rainfall for high rainfall (1100 mm mean annual rainfall) and 20% of annual rainfall for low rainfall areas (900 mm annual rainfall).

Thinning of high rainfall experimental catchments has led to maximum increases ranging from 8 to 18% of mean annual rainfall (90 to 200 mm), depending on the level of reduction of vegetation cover and catchment characteristics. The subsequent recovery of vegetation has led to water yields returning to pre-disturbance levels after 12 to 15 years.

Bauxite mining and rehabilitation within high-rainfall catchments has resulted in an increase of 8% water yield followed by a return to pre-disturbance levels after 12 years. There is uncertainty whether the water yield from the catchments with bauxite rehabilitation will continue to decline to below pre-disturbance levels.

Reforestation has been observed to lead to reductions in water yield which stabilise ten years after replanting.

Long-term predictions for water yield from areas subject to forest thinning, timber harvesting and regeneration, and bauxite mining are uncertain due to the complex inter-relationships between vegetation cover, tree height and age, and catchment evapotranspiration. Management of forests for water yield will need to acknowledge this complexity and evaluate forest management strategies at both the large catchment scale and at long time scales.

The extensive network of small catchment experiments, regional studies, process studies, and catchment modelling at both small and large scale important and integral components of research to develop management strategies to optimise water yield from the jarrah forest without forfeiting the environmental, social and cultural forest values.

1 Introduction

1.1 Purpose and scope

The population of the Perth–Bunbury region in Western Australia is currently 1.1 million, and is predicted to reach 3.1 million by 2050 (Western Australian Planning Commission 2000). At present, about 50% of Perth's water supply comes from surface water resources in the northern jarrah forest. Increasing demand has generated pressure to manage existing water sources better and to develop new sources more efficiently. Since the European settlement of Western Australia, land use practices in the jarrah forest have changed water yield and caused some serious water quality problems. The most significant activities are clearing for agriculture, forest harvesting and regeneration, and bauxite mining. Forest thinning to increase water yield has the potential to affect water yield and water quality.

1.2 Geomorphological description

The main physiographic feature of Western Australia is the Great Plateau (Mulcahy et al. 1972). Near Perth, the Plateau is separated from a sedimentary basin (called the Swan Coastal Plain) by the Darling Scarp, an almost linear north-south escarpment up to 300 m high. The adjacent margin of the Plateau forms part of a stable Archaean shield composed largely of granite which has invaded linear belts of metamorphosed sedimentary and volcanic rocks, some isolated occurrences of which remain. Thin sheet-like dolerite intrusions occur abundantly in the basement rock, and are known to affect groundwater movement (Engel et al. 1987). Close to the Scarp, there are deep V-shaped valleys with shallow soils and frequent rock outcrops. Further inland, these valleys transform to broad V-shaped valleys and then to very broad valleys with long (1–2 km) sideslopes of low inclination (less than 10%). The inland areas are characterised by profiles that have been deeply weathered in situ (typically 20–50 m). In inland areas the drainage is ephemeral and sluggish, whereas near the Scarp streams are often perennial and efficient. The greater incision near the Scarp is a result of the westerly drainage through this tectonically uplifted area.

1.3 Climate

The climate of the jarrah forest is conventionally described as Mediterranean. Further inland from the Darling Scarp in the north and from the coast in the south, the mean annual rainfall decreases markedly (Fig. 1). Along this gradient mean annual evaporation increases and streamflow decreases. In the lower rainfall areas of the forest (< 700 mm mean annual rainfall) virtually all the rainfall is evaporated, resulting in the accumulation of salt in the soil (Schofield & Ruprecht 1989). Salt accumulation does not occur to the same extent in the higher rainfall areas.

Hydrological characteristics, such as average water yields and annual peak flows, and levels of soil salt storage, all vary significantly with annual rainfall. For this reason, three rainfall zones are recognised in this report. They are high (average annual rainfall greater than 1100 mm), intermediate (average annual rainfall between 900 and 1100 mm) and low rainfall (average annual rainfall less than 900 mm).

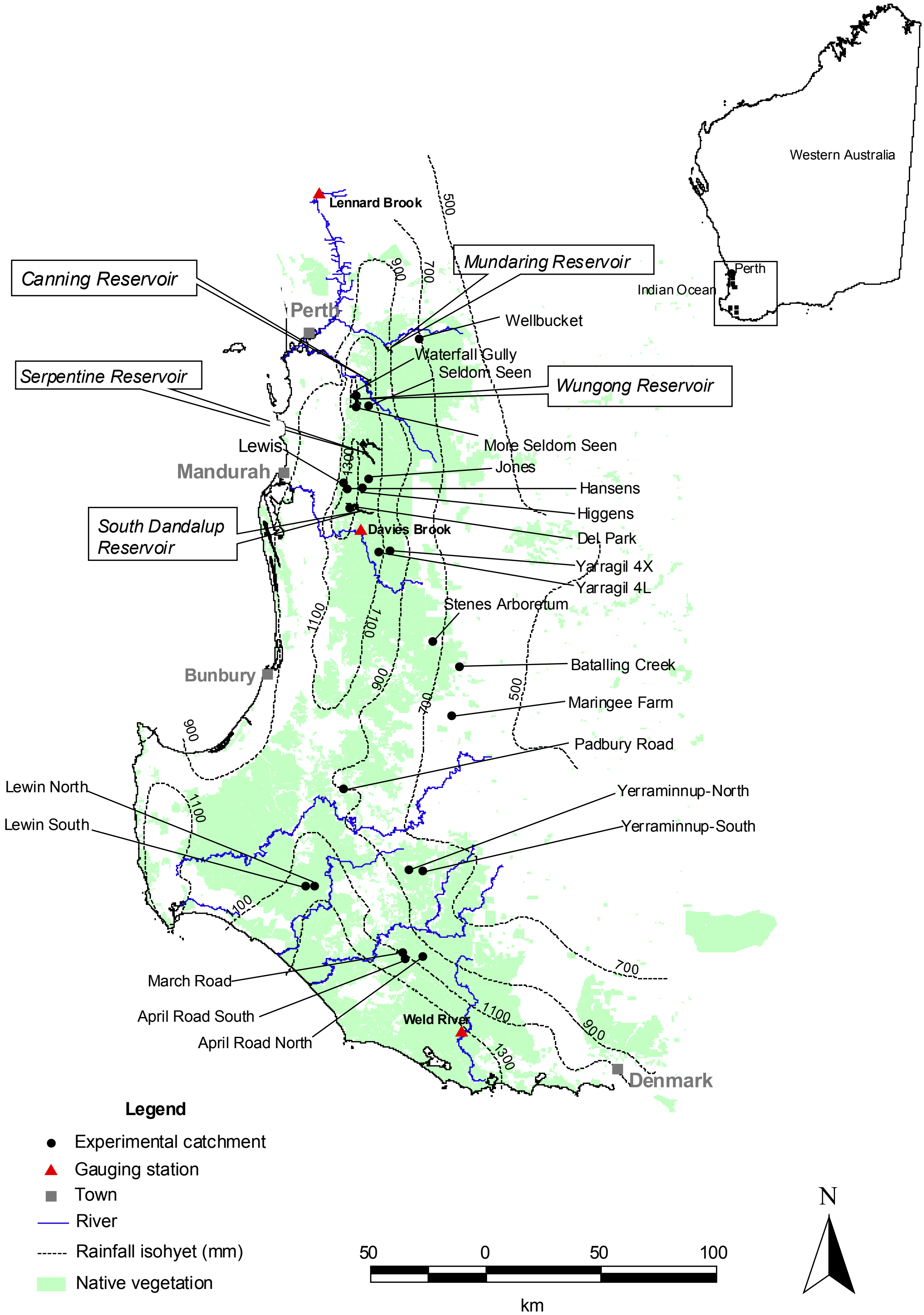

Figure 1. Location of experimental catchments

2 The jarrah forest and current disturbances

The jarrah forest forms part of the evergreen forest of Western Australia (Ovington & Prior 1983). As the rainfall decreases towards the north and east, the trees decrease in height thus forming a woodland or low forest. The jarrah forest is dominated by jarrah (*Eucalyptus marginata*), marri (*E. calophylla*), and other tree species including yarri (*E. patens*). There is an understorey of small trees and a groundcover of woody shrubs.

2.1 Forestry operation

There has been timber logging in Western Australia for more than 100 years. During this period, various methods of logging and subsequent regeneration have been practised (Borg et. al 1988). Logging practices have varied considerably according to location, forest, quality and the forest management practices of the day. Before 1920, large areas of the Northern Jarrah Forests were clear-felled and regenerated. During the period of 1920 to 1940, a group selection silvicultural system was used. In the period 1940–60, group selection cut and other areas had been subject to single tree selection cutting. In the 1960s, the logging strategy was changed from light selection cutting to heavy selection cutting and clear-felling, specially in the Southern forests. As a result of the varying logging practices, the jarrah forest now consists of various ages and structures of regeneration.

Jarrah dieback disease could have significant impact on the water yield from the forested catchments (Schofield et al. 1989). It is a plant disease caused by the soil-borne fungus *Phytophthora cinnamomi* (Podger 1968). The fungus is an introduced species and has the potential to be spread through forestry operations. The dieback disease is more prevalent in the Northern Jarrah Forest and covers approximately 14.2% of the area (Davidson & Shearer 1989).

2.2 Bauxite mining

Bauxite mining is a major land use within the Northern Jarrah Forest of Western Australia, with typically 500 ha cleared annually. By 2002, about 3000 ha had been cleared for mining of which 90% had been rehabilitated (Alcoa World Alumina Australia 2003). The principal bauxite mining area covers 50–60% of the Northern Jarrah Forest and most of the water supply catchments.

When bauxite mining started in the 1960s, its effects on water resources in terms of salinity and turbidity were expected to be negligible. Nevertheless, due to increased public concerns about the effects of mining on water resources and the jarrah forest environment, a Joint Intermediate Rainfall Zone Research Program (JIRZRP) was developed in the 1970s. This is a long-term ongoing research program with major activities scheduled until at least 2017 and has two main objectives:

i) to determine what impact bauxite mining will have on the water resources of the region and

ii) to document the forest, mine and rehabilitation management practices.

2.3 Clearing and agriculture

Agricultural development has involved clearing large expanses of native vegetation for cereal crops and pastures. The development of agriculture was slow until 1900 but expanded rapidly from 1900–30 and again from 1955–85. The early development was characterised by valley floor and lower slope forest clearing on the more fertile and wetter soils. From the 1950s, agricultural development expanded the early cleared areas upslope from the valley bottoms towards the ridges, as well as opening up new land in increasingly lower rainfall areas.

2.4 Experimental catchments network

In the 1970s, 27 experimental catchments in five groups were established in the south-west of Western Australia. The primary objective was to understand the effects of land use changes on streamflow, water quality and especially salinity. Over the last two decades, intense monitoring of hydrological parameters has led to better understanding of the water yield issues and the salinity generation processes. The catchments in the five land use groups are shown in Figure 1 and Table 1.

- Forest clearing for agricultural development
- Forestry operation
- Forest thinning for water production
- Bauxite mining and rehabilitation
- Reforestation

In 1974, a set of five experimental catchments was established in the Collie River catchment. The main objective was to quantify the effects on streamflow and salinity of typical clearing of native forest for agricultural development (Fig. 1). In 1977, different proportions of the vegetation cover of three catchments (Wights, Lemon, Dons) were cleared for pasture development (Table 1).

Seven research catchments were established in the Southern Forest during the 1970s to quantify the effects of operational logging practices on stream yield and salinity (Fig. 1). Streamflow, stream salinity, groundwater level and groundwater salinity were monitored before, during and after different ways of logging, and subsequent regeneration. Lewin South, March Road, April Road North and Yerraminnup South catchments underwent different forest management strategies and the others remained as controls (Table 1). Lewin North was the control for Lewin South, April Road South control for both March Road and April Road North, and Yerraminnup North control for Yerraminnup South.

Seven catchments were established in the Northern Jarrah Forest to understand the effects of different scales of forest thinning on stream yield and salinity (Fig. 1). Detailed analyses of the experiments were reported by Robinson et al. (1996) and Moulds et al. (1994). All of these catchments had jarrah–marri forests that, before the forest thinning program, had regenerated from selective logging during the 1940s and 1950s. Lewis and Hansens had, respectively, 26 and 27% dieback-affected areas, while Jones had only 6% affected area and Higgens was unaffected (Public Works Dept. 1984).

Another set of experimental catchments was established within the Northern Jarrah Forest under the Joint Intermediate Rainfall Zone Research Program (JIRZRP). Most of these experimental catchments are jointly funded and operated by World Alumina Australia (Alcoa) and the Water and Rivers Commission, now part of the Department of Environment. The research program is described in detail by Croton and Dalton (1999). The principal objective of the program was to understand the effects of bauxite mining and rehabilitation on catchment water yield and salinity. The experimental catchments are located in the High and Intermediate Rainfall Zones. More than 20 years of data are available from Del Park, Seldom Seen and More Seldom Seen catchments (Table 1).

Between 1977 and 1983, four experimental sites were established in the south-west of Western Australia to quantify the effects of reforestation techniques on groundwater level, streamflow and salinity. Those sites are within the Collie and Helena River catchments. Four reforestation methods with variable stem densities and location in the landscape were tested (Bari 1998). Three experimental catchments were established with different proportions of the cleared area replanted to quantify the effects of reforestation on streamflow and salinity load (Table 1).

Table 1. Summary of experimental catchments

Land use	*Catchment*	*Area (ha)*	*Rainfall Zone*	*Treatment*	
				Year	*Method*
Clearing for agriculture	Wights	94	High	1977	Clearing for agricultural development — 100% catchment cleared
	Salmon	82	High		None – Control
	Ernies	270	Low		None – Control
	Lemon	344	Low	1977	Clearing for agricultural development — 54% catchment cleared
	Dons	350	Low	1977	Block and park land — 38% catchment cleared
Forest harvesting and regeneration	Lewin South	90	High	1982	Heavy selection cut, regenerate naturally — no stream buffer
	Lewin North	113	High		None – Control
	March Road	261	Intermediate	1982	Clear felled and then replanted — no stream buffer
	April Road North	248	Intermediate	1982	Clear felled and then replanted — stream buffer retained
	April Road South	179	Intermediate		None — Control
	Yerraminnup South	183	Low	1982	Heavy selection cut and regenerate naturally — stream buffer retained
	Yerraminnup North	253	Low		None — Control
	Wellbucket	465	Low	1977	Selective logging and regenerate naturally
Forest thinning	Lewis	201	High		None — Control
	Hansens	78	High	1986	Uniform thinning
	Higgens	60	High	1986	Uniform thinning
	Jones	69	High	1989	Operational thinning
	Yarragil 4X	273	Intermediate		None — Control
	Yarragil 4L	128	Intermediate	1983	Operational thinning
Bauxite mining and rehabilitation	Del Park	131	High	1975–89	Bauxite mining and rehabilitation
	Seldom Seen	753	High	1966–97	Bauxite mining and rehabilitation
	More Seldom Seen	327	High	1968–95	Bauxite mining and rehabilitation
	Waterfall Gully	874	High		Control
Reforestation	Padbury Road	170	Low	1977	74% of the cleared area planted
	Maringee Farm	1299	Low	1982	18% of the cleared area planted
	Batalling Creek	1660	Low	1985	38% of the cleared area planted
	Stenes Arboretum	84	Low	1979	100% of the cleared area planted

3 Hydrological characteristics

3.1 Rainfall and evapotranspiration

Rainfall in the south-west of Western Australia is highly seasonal, with more than 80% falling between May and October, that is, during the winter months. Annual rainfall varies from more than 1400 mm on the western boundary to 600 mm on the eastern boundary of the jarrah forest (Fig. 1). Rainfall intensity is typically low and rarely exceeds the infiltration capacity of the undisturbed forested top soil (Sharma et al. 1987; Ruprecht & Schofield 1989).

Evapotranspiration is a major component of the hydrological cycle in the south-west. Sharma (1983) estimated that about 90% of the annual rainfall is lost through evapotranspiration, while Williamson et al (1987) estimated that about 75–80% of the annual rainfall is transpired by trees. Annual average interception by mature jarrah forest has been calculated to be about 15% of annual rainfall (Stokes 1985; Williamson et al. 1987). A recent data review of the interception experiment conducted at the Del Park catchment shows that 13% of annual rainfall is lost though interception alone (Croton & Norton 1998). The mature jarrah forest extracts water for transpiration from a considerable depth of the soil profile, and sometimes from the deeper permanent groundwater (Carbon et al. 1980). Marshal et al. (1994) measured annual tree water use of jarrah stands at the Del Park and Hansen catchments. Average annual tree water uptake was 485 mm, approximately 34% of annual rainfall. Greenwood et al. (1985) used the ventilated chamber technique to measure evapotranspiration from the middlestorey and ground layer at Del Park catchment. The middlestorey trees evaporated 16% and ground layer 37% of rainfall respectively.

3.2 Streamflow generation

The sources of streamflow are overland flow, throughflow and groundwater flow. The soil characteristics of the study region favour throughflow as the major source of streamflow. The soils generally comprise highly permeable sandy to loamy A horizons overlying clays of low permeability at 30 to 100 cm depth (McArthur & Clifton 1975). During winter, rain infiltrates the soil, perches on the clay B horizon and flows downslope to discharge to streams. Over 90% of streamflow may be generated by this shallow throughflow (Stokes & Loh 1982). However, on an average annual basis, only 25% of rainfall in the high rainfall areas becomes streamflow and, in the low rainfall areas, this value is as low as 1%. Only a very small portion of rainfall recharges groundwater and most of the rainfall is lost by evapotranspiration. A significant proportion of winter rainfall can be stored in the soil and transpired or evaporated during spring and summer.

Overland flow is usually restricted to low landscape positions that become saturated. According to Stokes and Loh (1982), these variable source areas contribute only about 5% of the annual flows, but dominate the generation of peak flood flows.

3.3 Groundwater and salt discharge

Groundwater is conveniently described by two systems — the perched aquifer and the deeper (or permanent) aquifer. The perched aquifer is ephemeral and, where it does not intersect the permanent groundwater, is fresh. Where saline permanent groundwater has contributed to the perched aquifer, shallow throughflow may carry substantial amounts of salt to the stream.

The permanent groundwater system in the high rainfall areas (annual rainfall above 1100 mm) usually discharges to streams and keeps these areas leached of salt. As such, the high rainfall areas are generally of low salinity hazard. In the low rainfall areas, the deep groundwater tables are far below the stream bed and do not contribute to streamflow. As a consequence, salt has accumulated in the unsaturated zone and these areas pose a high potential salinity hazard. The deep unsaturated zone means that only persistent forest changes which increase recharge over long time periods (tens of years) are likely to cause long-term stream salinity problems. These conditions have occurred in the case of agricultural development.

Partial or transient land use changes in the intermediate rainfall areas (annual rainfall between 900 and 1100 mm) often pose a major problem because the permanent groundwater is moderately saline but the watertable is not far below the streams.

3.4 Stream salinity characteristics

Stream salinity is governed by the relative volumes and salinities of overland flow, shallow throughflow and groundwater flow. As described above, permanent groundwater is the major source of salts, and throughflow and overland flow tend to dilute this source.

Annual mean stream salinities of forest catchments range from 80 to 400 mg/L Total Dissolved Salts (TDS). The lowest stream salinities (~ 100 mg/L TDS) occur in the low rainfall areas where groundwater does not discharge to streams (Schofield & Ruprecht 1989). In the high rainfall areas, where discharging groundwater salinities are low, stream salinities are again low (~ 150 mg/L TDS). Streams in the intermediate rainfall area have the highest salinities in all forest areas. Annual average values are commonly 250 mg/L TDS, but can approach 400 mg/L TDS. These high salinities result from a limited discharge of groundwater of moderate salinity combined with throughflow of low salinity.

Stream salinity can also show strong seasonal and annual variations. Seasonally, the variations are usually associated with the proportion of throughflow, which is high in mid-winter and low in spring and autumn. Annual variability may be attributed to annual variations in throughflow due to rainfall variation and to longer-term changes in groundwater level resulting in greater or smaller contributions of groundwater to streamflow.

3.5 Stream sediment concentrations

Stream sediment concentrations of undisturbed forest are generally below the level of reliable detection (5 mg/L suspended solids less than 63 microns in size). The highly permeable soils and related lack of surface runoff, the generally low relief, the high amounts of litter and understorey vegetation all contribute to low sediment loads. Organic material from the natural decay of vegetation and disturbance due to logs falling naturally into stream channels have been the major sources of suspended material measured in undisturbed catchments. Roads constructed for moving machinery and transport vehicles also play some role in generating sediment loads.

4 Impact of climate variability

Climate variability is a major concern for many people in Western Australia. The run of below-average rainfall observed in many areas between 1975 and the present has led to below-average streamflow in many rivers.

4.1 Rainfall

Rainfall in the south-west of Western Australia declined systematically over the last 100 years. Plots of annual rainfall with time show a consistently low rainfall particularly in the 1980s and into the 1990s. Perth, Albany and Bunbury demonstrate this consistent low rainfall period (Fig. 2).

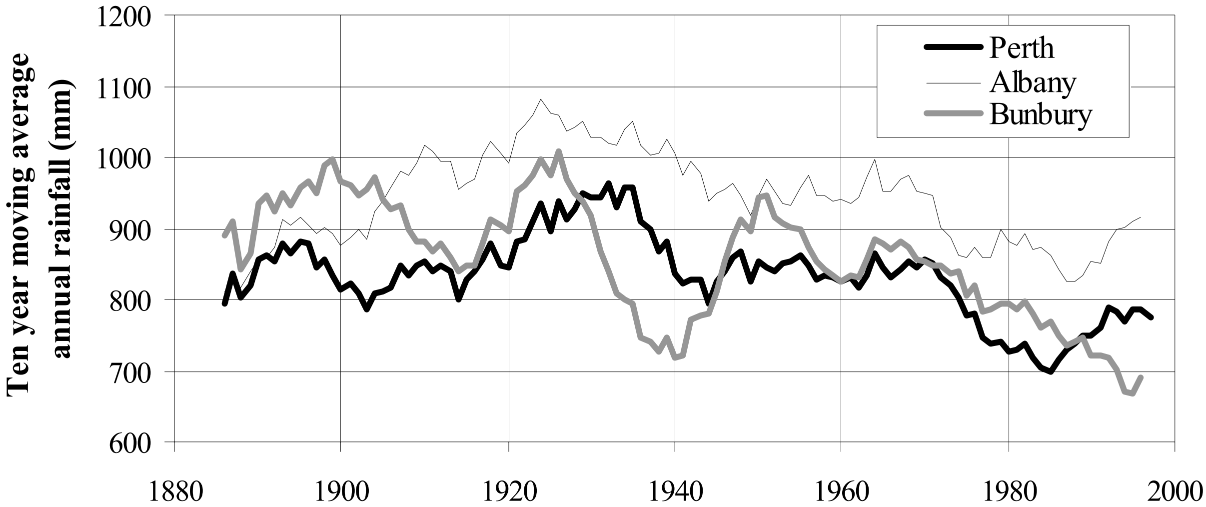

Figure 2. Annual rainfall variability over the last 100 years

The reductions in rainfall observed from 1910 to 2001 are not uniform across the south-west. The reduction has been most significant in parts of the south-west (Fig. 3). Average May to October rainfall in the south-west over the last 25 years has been only 85–90% of the preceding 50-year average. In the central Wheatbelt area, average May–October rainfall increased slightly (Fig. 3).

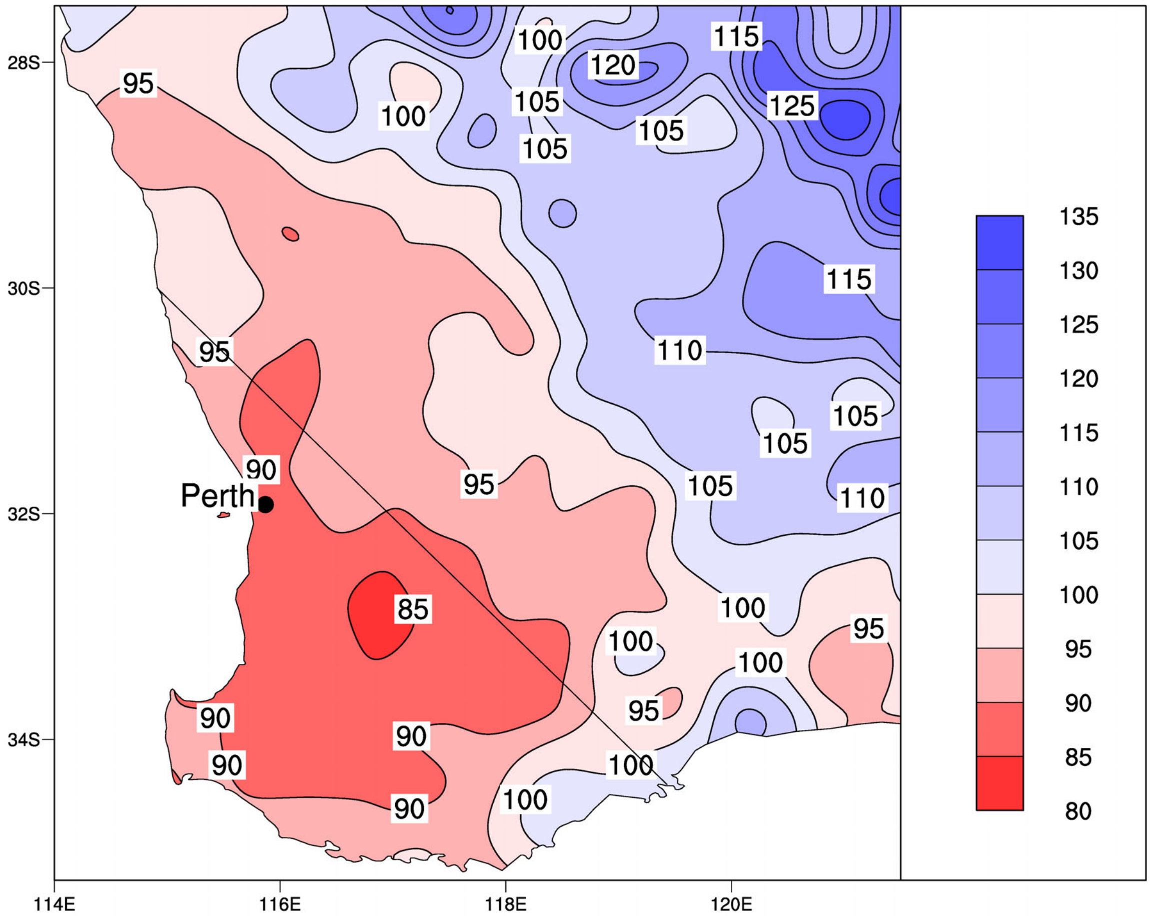

Figure 3. Average May–October rainfall over the 1976–2001 period as a percentage of the average May–October over the 1925–75 period (After IOCI, 2002)

4.2 Streamflow

The impacts of climate variability on annual totals of river flow were studied using data from Water and Rivers Commission gauging sites and modelled data at water supply reservoirs for Perth.

Fifteen of the 33 rivers analysed had statistically significant trends of decreasing streamflow and three had statistically significant trends of increasing streamflow. The rivers with increased streamflows had major areas of clearing within their catchments.

Two examples of the variation in annual streamflow in Figure 4. Annual streamflow in the Weld River along the south coast fell from 75 000 gigalitres (GL) to slightly below 40 000 GL (Fig. 4a). Annual streamflow in the Davies Brook, a tributary of the Murray River, fell from about 10 000 to 4000 GL (Fig. 4b). The lower flows during the period 1975–2000 are clearly shown.

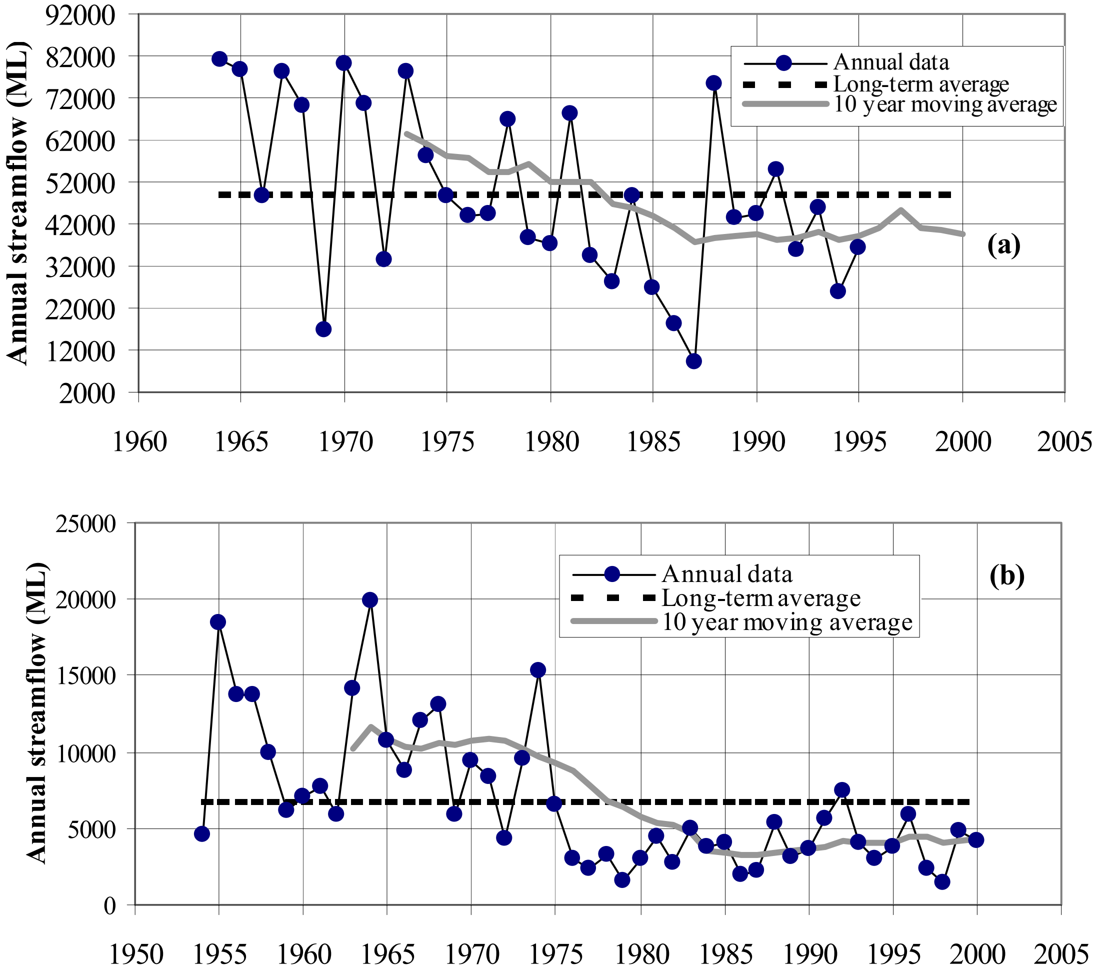

Figure 4. Trends in annual streamflow for two rivers in the south-west (a) Weld River and (b) Davies Brook

4.3 Perth water supply catchments

The estimated data for the Perth water supply reservoirs typically commence in 1911 and also indicate the trend to lower flows since the mid-1970s (Fig. 5). There are no large river flows over the last 25 years.

The combined mean annual inflow of the metropolitan sources for the period 1912–2000 is more than 290 GL (Fig. 5). The combined mean annual inflow for 1975–2000 is 172 GL, a 41% reduction in the long-term mean (1912–2000). This indicates there has been an extended period of below-average annual inflows to the metropolitan reservoirs since about 1975 (Fig. 5).

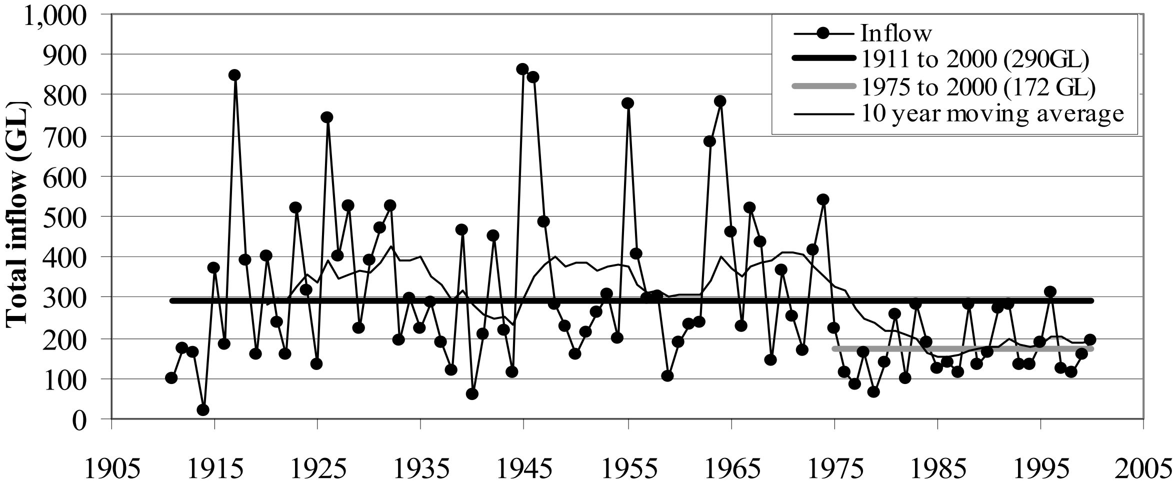

Figure 5. Yearly inflow into Perth water supply reservoirs

The reduction in inflow for the period 1975–2000 compared with the 1911–2000 mean was not uniform for all reservoirs. The mean inflow to the Serpentine Reservoir for 1911–2000 was 66 GL and, for 1975–2000, it was 45 GL, which is a 32% reduction (Fig. 6a). The mean inflow to the Canning Reservoir during the 1975–2000 period was 27 GL, so the mean inflow was 46% lower than the 1911–2000 mean inflow of 57 GL (Fig. 6b). As expected, catchments in the lower rainfall areas are more affected than those in higher rainfall areas.

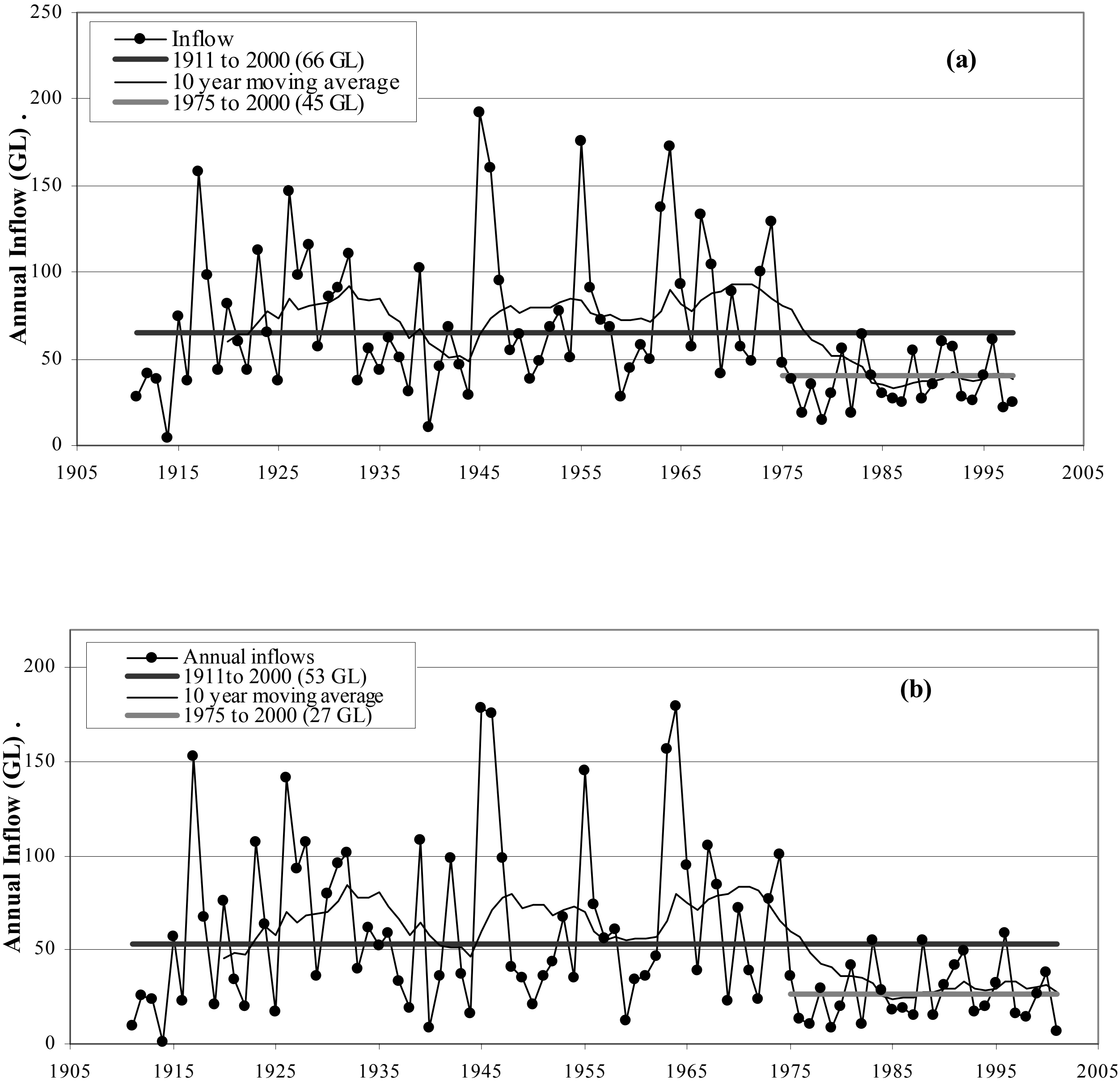

Figure 6. Modelled and observed data for inflows into the (a) Serpentine and (b) Canning reservoirs

5 Effects of forest disturbance on water yield

5.1 Clearing forest for agricultural development

Forest clearing for agricultural development generally results in rising groundwater level and increased streamflow volume and discharge. The rate of rise depends upon the environmental settings of the catchment. The rate of groundwater level rise decreases with decreasing annual rainfall so groundwater takes longer to reach the streambed. Stream salinity increases dramatically with decreasing rainfall once the groundwater level reaches the streambed. It is clearly seen in the data collected from the set of five experimental catchments in the Collie River catchment (Fig. 1).

Analyses of the data show that the groundwater level rose in both Wights and Lemon catchments. Before clearing, the permanent groundwater system was within a few metres of the stream zone of Wights catchment (Ritson et al 1995). After clearing in 1977, the groundwater level rose and reached a new equilibrium by 1984. At the Lemon catchment, the permanent groundwater system was some 20 m below the streambed and did not initially play any role in streamflow generation. The groundwater level rose after clearing in 1977 and reached the streambed by 1987 (Croton & Bari 2001). Application of a fully-distributed catchment model revealed that the groundwater system reached a new equilibrium by 1996 (Croton & Bari 2001). When the groundwater system reaches the streambed it creates ‘permanently’ saturated areas along the stream zone. Observations of satellite photographs show that about 18% of Wights catchment was saturated when it reached its new groundwater equilibrium by 1984 (Bari et al. 2003) but only about 8% of the Lemon catchment area was saturated when its groundwater reached equilibrium in 1996.

Analyses of streamflow data show that there were two phases of streamflow rise after clearing of native forest at the Wights and Lemon catchment (Ruprecht & Schofield 1989; Bari et al. 2003). There was an immediate increase in streamflow due to the reduction in evapotranspiration. The second phase (which was more prominent at the Lemon catchment) occurred when the groundwater level reached the streambed. These phases are clearly visible from the annual rainfall–streamflow relationship for Lemon catchment (Fig. 7). Annual streamflow increased from 17 to 150 mm for a given rainfall of 750 mm when the catchment achieved its new hydrological stability. Compared with the control catchment, annual streamflow at the Lemon catchment increased 15 times whereas, at the Wights catchment, the increase was only four-fold. The average increase water yield at the Wights catchment was 32% of annual rainfall compared with 17% at the Lemon catchment (Fig. 8).

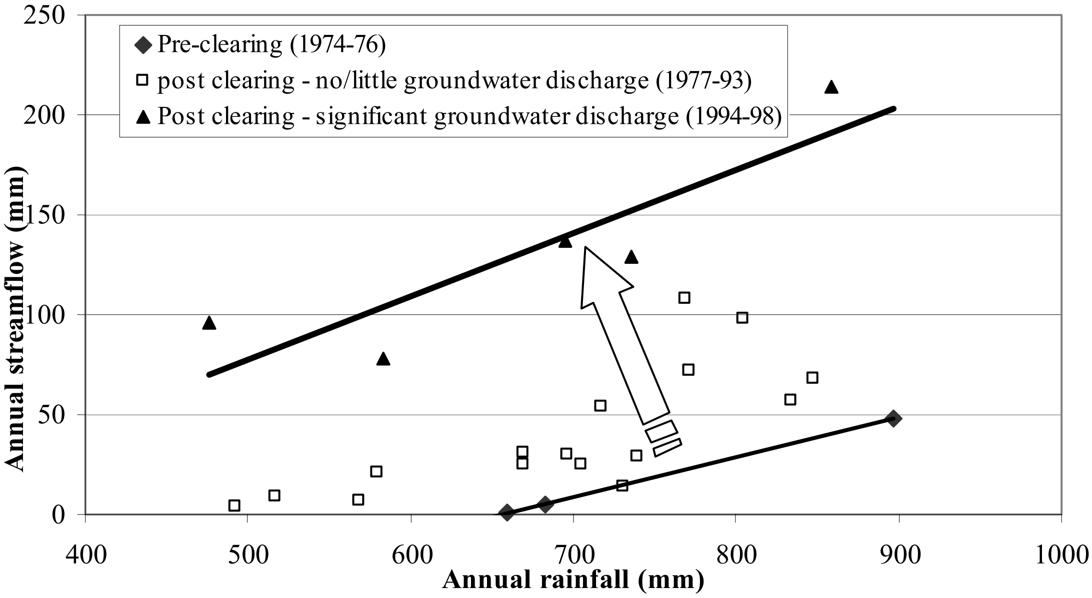

Figure 7. Annual streamflow relationship with rainfall changing with time

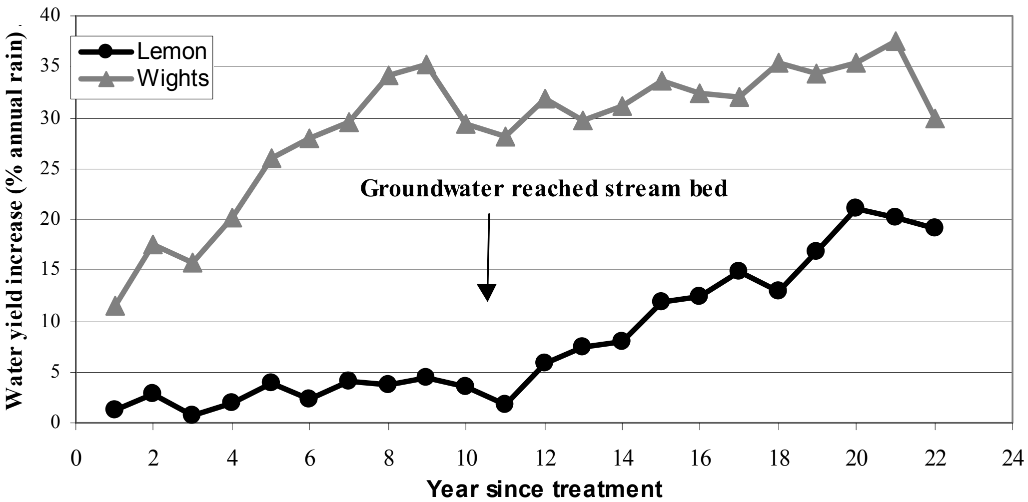

Figure 8. Increase in water yield due to clearing for agriculture

Salt storage in the soil profile at the Wights and Lemon catchments is 0.4 kg/m^2 and 2.3 kg/m^2 respectively (Johnston 1987; Bari et al. 2003). The annual stream salinity at the Wights catchment increased immediately after clearing from an average of 360 to 515 mg/L TDS (Fig. 9). The average annual salt load increase at the Wights catchment was 14-fold (compared with the control catchment). At the Lemon catchment, from 1977–87 (before the groundwater system was connected to the streambed), the average annual stream salinity rose from 80 to 127 mg/L TDS. After 1987 when the groundwater system reached the streambed, the stream salinity generation process changed significantly and annual average salinity increased systematically to 1700 mg/L TDS. The annual stream salt load increased 180-fold compared with the control catchment. Recent modelling shows that under present land use and climate conditions, the annual stream salinity of the Lemon catchment will be stable for at least the next 50 years (Croton & Bari 2001).

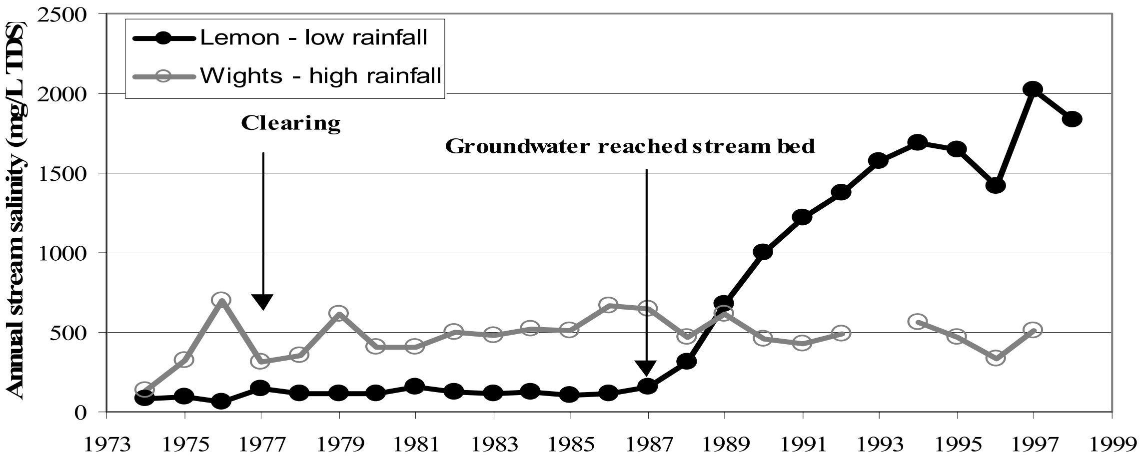

Figure 9. Increase in stream salinity due to clearing for agriculture

5.2 Forest harvesting and regeneration

Experiments on the effects of forest harvesting and regeneration on stream yield were conducted at eight catchments in the south-west of Western Australia (Fig. 1, Table 1). The mean annual runoff from the three control catchments varied from 17.2 to 3.2% (Table 2). All treated catchments started flowing in April/May after significant rainfall and stopped flowing in November or early December. After logging, streamflow and flow-duration increased at most of the treated catchments (Bari & Boyd 1993). The treated catchments started to flow earlier than the control catchments. The highest average streamflow was recorded at the Lewin South catchment and the lowest at the Yerraminnup South catchment recorded the lowest. The treated catchments showed significant increases in flow volumes and peak flow compared with the control catchments.

The regression equations developed between the treated and the control catchments were based on the pretreatment periods and were used to estimate annual streamflows which would have occurred if harvesting and regeneration had not taken place. The differences between the estimated and the observed streamflows were considered to be the effects of treatments. Annual streamflows increased for two to three years after logging and then began to decline (Fig. 10). The maximum increase was about 15% of annual rainfall at the Lewin South, March Road and April Road North catchments while for the Yerraminnup South catchment the increase was only about 5% . By 1996 (14 years after treatment), the streamflow of all the treated catchments (except April Road North) seemed to have returned to pretreatment levels (Fig. 10).

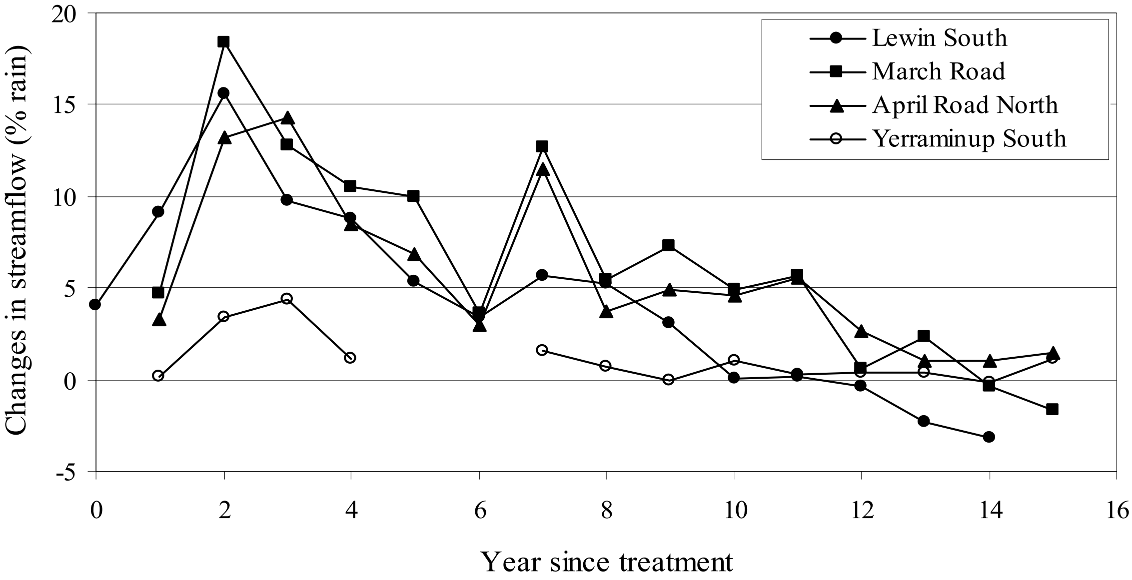

Figure 10. Streamflow response to logging experimental catchments in the Southern Forest

5.2.1 Salinity

The changes in stream salinity and salt load were evaluated from the relationships between the streamflow, salinity and salt load at the treated catchments before and after logging. The method is described by Bari and Boyd (1993). After logging, stream salinity increased at all the treated catchments. The annual stream salinity at three treated catchments, but not the March Road catchment, remained below 500 mg/L TDS (the upper limit for fresh water resources set by the Western Australian Water Resources Council 1986) (Fig. 11). In the dry-year 1987, highest annual stream salinity (780 mg/L TDS) was recorded at the March Road catchment — the only record exceeding 500 mg/L TDS (Fig. 11). Most of the time the annual salinity was less than 200 mg/L TDS.

After logging, the annual stream salinity at Lewin South increased until 1985 and then levelled off. At March Road, stream salinity decreased in 1982 and then began to rise. The maximum increase in annual salinity (320 mg/L TDS) occurred in 1987. Since then there has been a decreasing trend (Fig. 12). If there had been no logging of the March Road catchment, annual stream salinity in 1987 would have been 460 mg/L TDS (Bari & Boyd 1993). At April Road North, stream salinity increased until 1987 and then began to fall. The maximum increase was 115 mg/L TDS. In 1994 the stream salinity was slightly higher than it would have been without logging. At the Yerraminnup South catchment there were no significant changes in stream salinity following logging (Fig. 12). After logging, the stream salt load increased at all treated catchments as a result of higher flow and salinity.

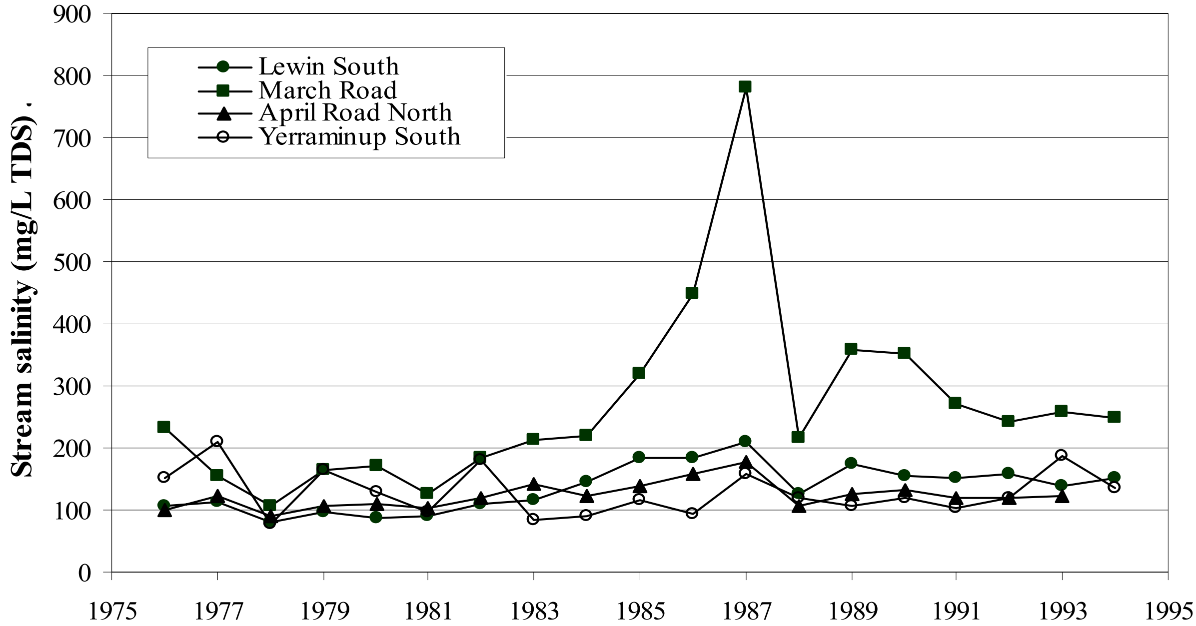

Figure 11. Annual stream salinity of four treated catchments

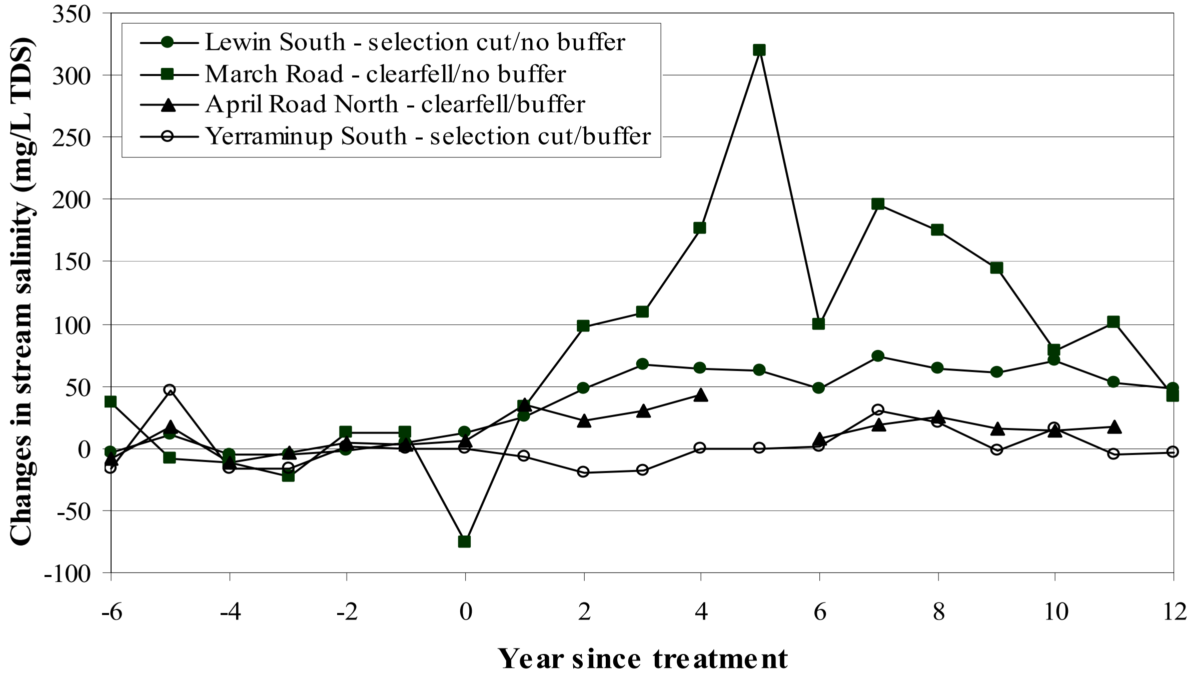

Figure 12. Changes in annual stream salinity in experimental catchments in the Southern Forest

5.3 Stream sediment concentrations

Before logging the mean annual stream sediment concentration in all seven experimental catchments was less than 5 mg/L. Sediment concentrations increased in the March Road and Lewin South catchments and remained elevated for one to three years (Fig. 13). These two catchments retained no buffer of streamside vegetation and were logged through both summer and winter. The highest annual flow-weighted sediment concentrations were 38 mg/L on the March Road catchment and 20 mg/L on the Lewin South catchment. Stream sediment concentrations are expected to return to pre-logging levels four or five years after logging (Fig. 13).

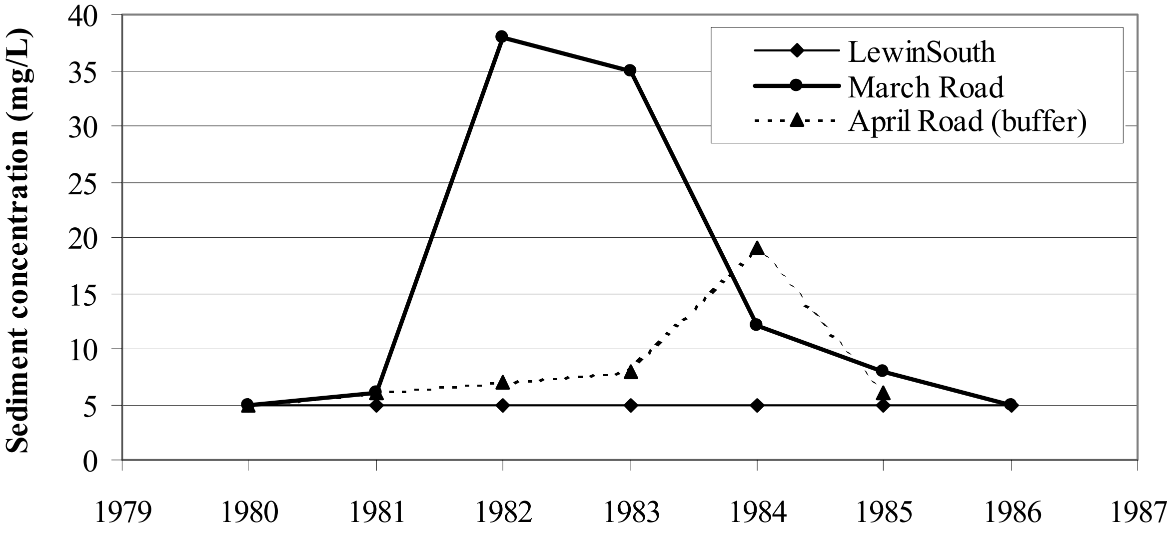

Figure 13. Changes in stream sediment concentration following logging in the Southern Forest

5.4 Forest thinning

Forest thinning generally results in an initial increase in streamflow followed by a return to pre-disturbance levels. The return to pre-disturbance levels can be delayed by regrowth suppression at the initial thinning or by later regrowth control. A paired-catchment method was used to quantify the effects of treatment. Mean runoff from the two control catchments (Yarragil 4L and Lewis) ranged from 8.2 to 1.5% of annual rainfall (Table 2).

The uniform and intensive thinning treatment which reduced crown cover on the Hansen catchment by 80% (from 60 to 14%) resulted in a maximum increase of 260 mm streamflow (19% of annual rainfall) after 3 years. Five years after treatment, there were systematic declines in streamflow (Fig. 14). Annual streamflow was approximately 90 mm higher than the pre-treatment flow 10 years after the forest thinning (Table 3). This declining trend in streamflow is attributable to forest regrowth following treatment (Robinson et al. 1997).

At the Higgens catchment, where the forest basal area was reduced by 60%, the mean streamflow during the pre-treatment period was 29 mm, about half of the Hansens' pre-treatment runoff rate. In the post-treatment period streamflow was considerably higher (Table 3). The increase in streamflow peaked and then declined steadily to its pre-treatment level within 10 years (Fig. 14).

At the Jones catchment, which was thinned to 40% of the basal area, the average post-treatment streamflow was 99 mm, which is about eight times the pre-treatment streamflow. The increase in streamflow peaked at 103 mm, five years after thinning then decreased throughout the remainder of the period (Fig. 14). Results from this catchment suggest that there has been a reduction in streamflow despite very little change in basal area since treatment (Robinson et al. 1997).

At the Yarragil 4L catchment, the treatment was to reduce basal area reduced from 35 to 11 m^2/ha —a reduction of 70%. The average pre-treatment streamflow was 3.8 mm and during the post-treatment period the average streamflow increase was more than 15-fold to 55 mm. The highest streamflow (95 mm or 11% of rainfall) was observed approximately nine years after treatment (Table 3). The post-treatment annual streamflow was still elevated 15 years after treatment (Fig. 15). These results emphasise the importance of ongoing management in order to sustain water yields.

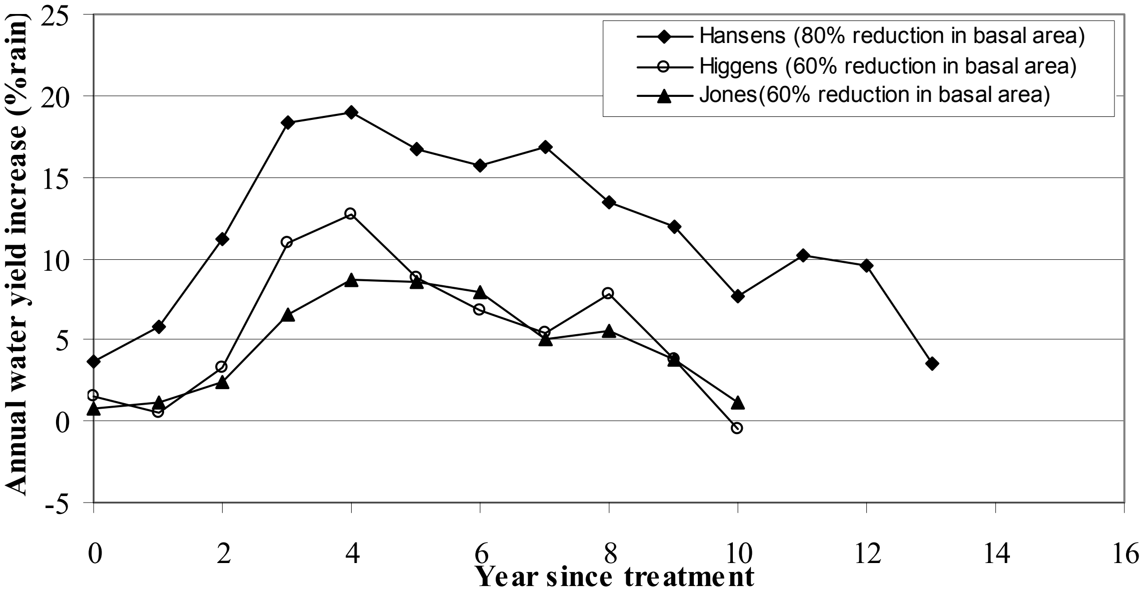

Figure 14. Water yield increases after forest thinning in high rainfall catchments

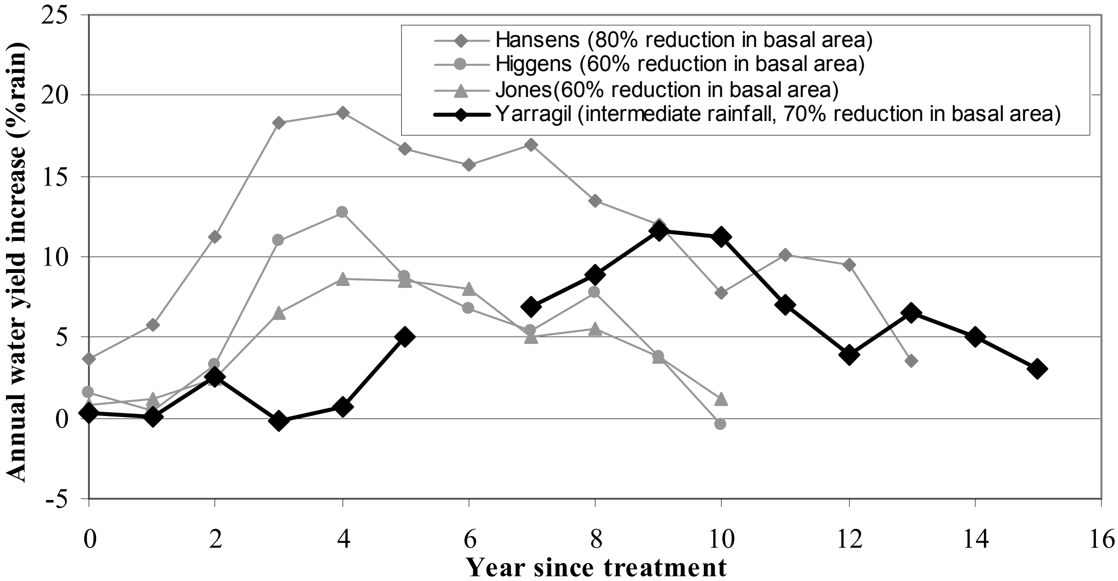

Figure 15. Water yield increases after forest thinning in low rainfall catchments

Table 2. Summary of water yield from forested control catchments

Catchment	*Rainfall*	*Average water yield*	
	(mm)	*(% rain)*	*(mm)*
Lewis	1190	8.2	99
Yarragil 4L	951	1.5	14
Waterfall Gully	1110	27.0	269
Lewin North	1100	17.2	189
April Road South	1012	10.7	108
Yerraminnup North	778	3.2	25
Salmon	1090	10.8	119
Ernies	712	0.7	5
Mean	**979**	**10.6**	**104**

Table 3. Summary of forest thinning impacts on water yield

Catchment	*Treatment (reduction in basal area) (%)*	*Maximum increase in water yield*		*Increase in water yield after 10 years*	
		(% rain)	*(mm)*	*(% rain)*	*(mm)*
Hansens	80	19	260	7.5	90
Higgens	60	12	156	0	0
Jones	60	9	103	0	0
Yarragil 4L*	60**	12	131	11	97

* Catchment thinned twice
** in crown cover

5.4.1 Salinity

The annual stream salinities measured are in the range of 110-120 mg/L TDS at the Hansens and Lewis catchments and 120-130 mg/L TDS at Higgens and Jones. These annual stream salinities are about the same as the groundwater salinities (Robinson et al. 1997). There were no rises in stream salinities in the ‘forest thinning for water production’ experiment. The streamflow on all catchments was fresh (well below 500 mg/L TDS) for the entire study period. These results are typical for streamflow in the High Rainfall Zone because of low salt storage (Robinson et al. 1997). At the Yarragil 4L catchment, in the Intermediate Rainfall Zone, the annual stream salinity was highly variable (from 75 mg/L to 170 mg/L TDS. At this catchment, there was no apparent increase in stream salinity following forest thinning, despite the relatively higher salt storage in the landscape, because there was no groundwater discharge to the stream.

5.5 Bauxite mining and rehabilitation

The impacts of bauxite mining and rehabilitation on streamflow have been monitored in detail at the Del Park, Seldom Seen and More Seldom Seen catchments. These three catchments are all in the High Rainfall Zone (Fig. 1).

The 'paired-catchment' method was used to quantify the effects of mining on water yield. The control catchment was Water Fall Gully, which had annual rainfall and pan evaporation higher than all three treated catchments and was also affected by dieback. Some of the streamflow records were faulty due to silting of the pools. Comparison with the control shows that, at the Del Park catchment, the streamflow increased a maximum of 8% of the annual rainfall (Fig. 16a) following mining and rehabilitation. There was a delay in the timing of the increase in annual streamflow and catchment area cleared for mining and not rehabilitated (Fig. 16a). When the rehabilitation was completed in 1989, water yield returned to the pre-mining level.

The gauging stations on Seldom Seen and More Seldom Seen catchments both started operating in April 1966 so effectively there were three years of streamflow data before mining operations on the More Seldom Seen catchment began but only two years on the Seldom Seen catchment. Data collected at the gauging stations suggest that, for both catchments, the streamflow was unaffected by clearing operations until about 1970 — so the four-year period 1966—1969 has been used to estimate the pre-mining relationship between the two mined catchments and Waterfall Gully control catchment. Fig. 16b shows an annual time series indicating streamflow increase as a percentage of rainfall, calculated as the difference between observed and predicted streamflow, divided by annual rainfall. The first four years of data (1966 to 1969) show streamflow increase consistent with the good fit of the regression line whereas, during years of clearing for mining, the streamflow increase reached a maximum value of 23% in 1981 and then declined steadily to between 0 and 10% after 1988. There was a very good correlation between the increase in streamflow and the percentage of the catchment area cleared for mining but not yet rehabilitated (Fig. 16b).

The results from the More Seldom Seen catchment were similar to the Seldom Seen catchment. Most years since 1970 plot above the regression line indicating that More Seldom Seen streamflow has increased relative to Waterfall Gully since mining began. The observed maximum streamflow increase of 21% in 1981 corresponded to approximately 21% clearing in that year (Fig. 16c). Since 1988, as rehabilitation of the catchment neared completion, the streamflow increase returned to values between ± 5%.

Recently a distributed catchment hydrology model has been applied to all three catchments (Boniecka et al. in prep.). The successful application of the model will better quantify the effects of rehabilitation on streamflow reduction.

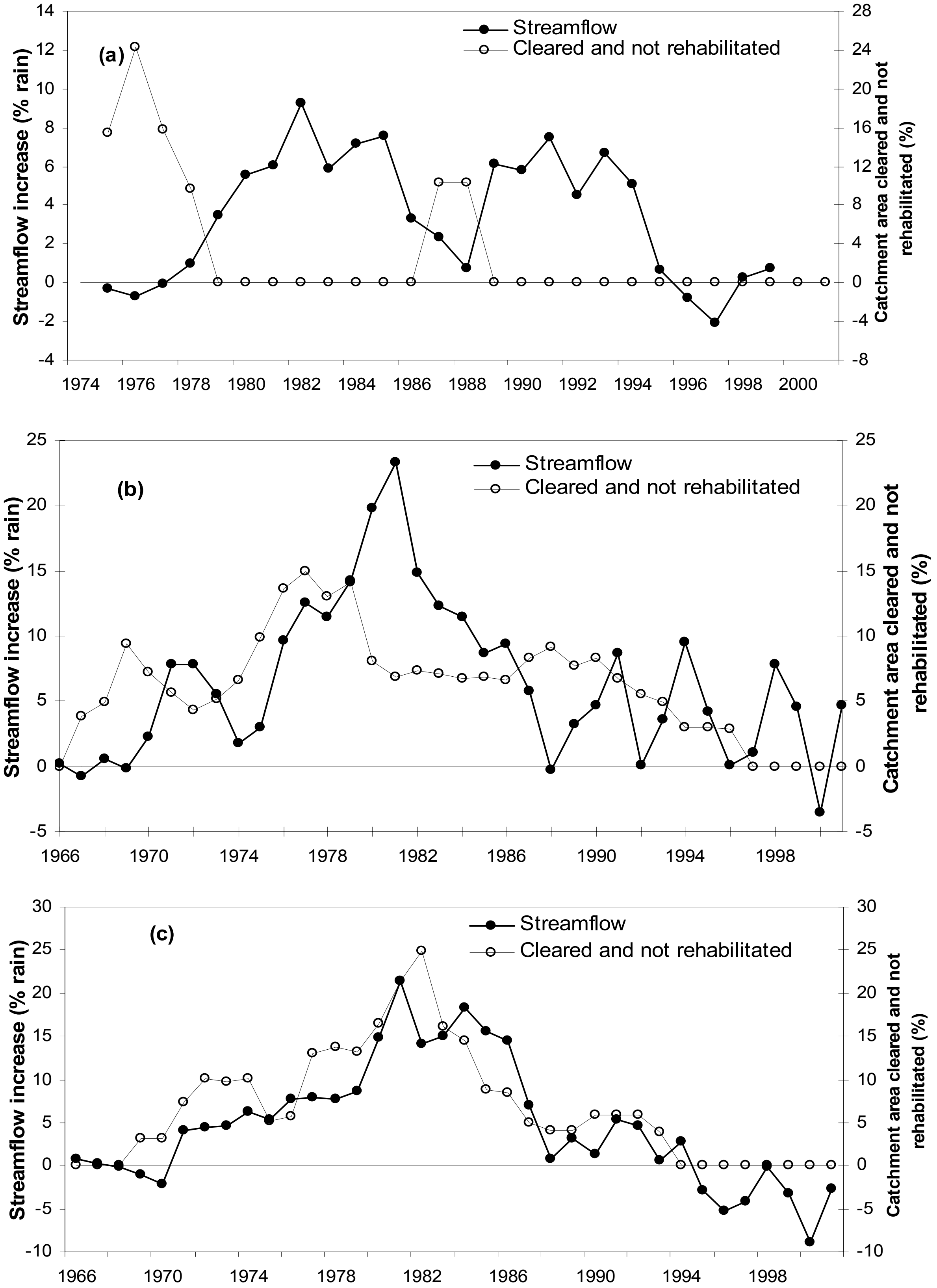

Figure 16. Changes in streamflow following bauxite mining and rehabilitation at (a) Del Park, (b) Seldom Seen and (c) More Seldom Seen catchments

5.5.1 Salinity

Stream salinity was measured by manual sampling at the Del Park, Seldom Seen and More Seldom Seen catchments. Annual stream salinities at the Seldom Seen and More Seldom Seen catchments were higher in the 1960s and late 1970s, but these resutlts may be biased by the small number of samples taken. Between 1976 and 2000 the annual salinities of these catchments were stable. Annual stream salinity at the Del Park catchment varied between 100 to 150 mg/L TDS and during most of the year was lower than at the other two catchments. The streamflow of all three catchments remained fresh (well below 500 mg/L TDS) during the entire study period.

5.6 Dieback

Dieback disease is believed to have a major impact on water yield of the Northern Jarrah Forest (Schofield et al. 1989). Studies found an increasing trend in streamflow associated with an increasing area affected by dieback in the Wungong water supply catchment (Batini et al. 1980). The reduced vegetation cover caused by dieback led to reduced transpiration and interception. Dieback sites are associated with soils with poor vertical drainage and greater throughflow (Shea et al. 1983; Kinal 1986). Average annual runoff rates from catchment with 'severe', 'some' and 'none' dieback-affected areas are listed in Table 4. Mean runoff from the 'severe' dieback-affected catchments was 20.5% compared with 9.1% from the 'some' dieback affected catchments (Table 4).

Table 4. Summary of the impact of dieback on water yield

Catchment name	*Rainfall*	*Runoff*		*Dieback status*
	(mm)	*(% rain)*	*Mean*	
Falls Brook	1230	17.3		Severe
Harvey River	1250	19.4	20.6	Severe
Waterfall Gully	1275	21.6		Severe
Tallanalla Creek	1110	21.6		Some
Davies Brook	1200	6.0		Some
Yarragil Brook	1075	2.9	9.1	Some
Pickering Brook	1050	5.9		None

5.7 Reforestation

The effects of reforestation techniques on groundwater level, streamflow and salt load have been reported previously (Schofield et al 1989; Bari & Schofield 1991; Schofield & Bari 1991; Bari & Schofield 1992; Bari & Boyd 1994; Bari 1998). Reforestation lowers groundwater level. Falls in groundwater level from 1 to 8 m (compared to the control areas) have been observed. The extent of watertable fall depends upon the proportion of the cleared area replanted and the stem density. If higher proportion of the cleared landscape is planted with higher stem density, a greater lowering in groundwater level is achieved. For example, at the Stenes Arboretum site, where almost all the cleared area was planted with an initial tree density of 1200 stems/ha, a maximum fall of 8 m has been observed. Ten years after reforestation, the groundwater system seemed to achieve its new stability and no further lowering was observed.

The effects of partial reforestation on streamflow and salinity were monitored at the Maringee Farm, Batalling Creek and Padbury Road catchments (Fig. 1). A fully-distributed catchment model was set up and applied to Maringee Farm and Batalling Creek. Results show that 18% reforestation at Maringee Farm, predominantly along the stream zone, reduced streamflow by 10% and salt load by 15% (Bari & Croton 2000). Salinity increased due to the disproportionate reduction in streamflow. It appears that, 10 years after reforestation, the catchment achieves a new stability and no further reduction in streamflow or load is to be expected (Bari & Croton 2000). Modelling of the Batalling Creek catchment showed similar results (Bari & Croton 2002).

The highest reduction in streamflow and salt load was observed at the Padbury Road catchment, where 75% of the cleared area was replanted between 1977–83 (Bari 1998). Annual streamflow and salt load fell to approximately 20% of the pre-planting values and stabilised in the 1990s, 10 years after planting. The average annual stream salinity increased from a pre-planting value of 1070 mg/L TDS to more than 2000 mg/L TDS in 1986–87 (Fig. 18). After that, there was a decreasing trend in stream salinity which stabilised at an average 1020 mg/L TDS in the mid-1990s (Fig. 18).

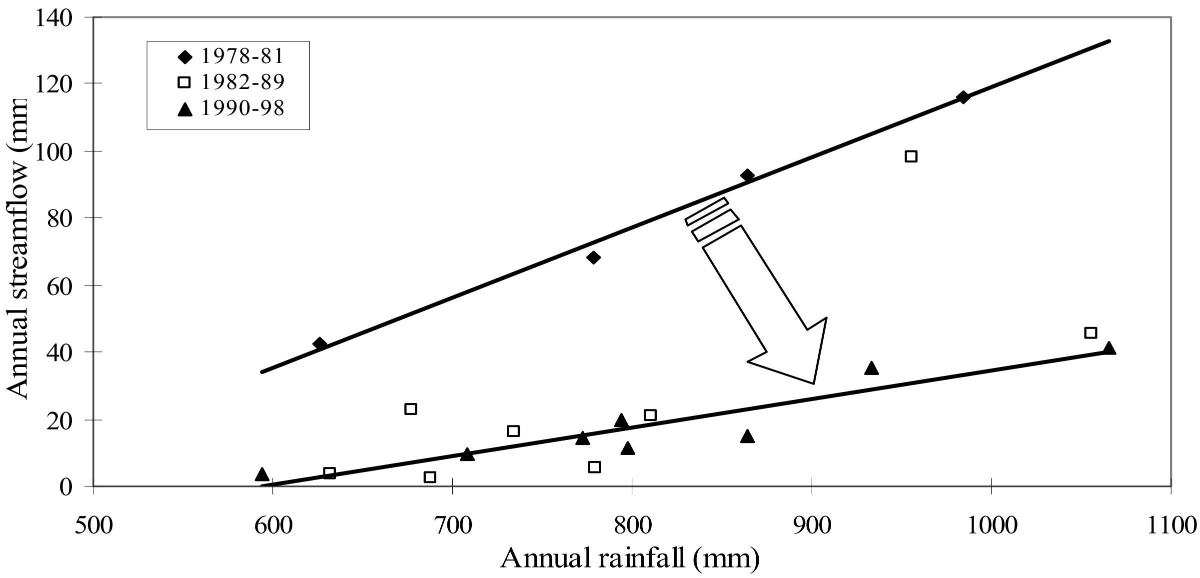

Figure 17. Relationship between annual rainfall and streamflow in the Padbury Road catchment

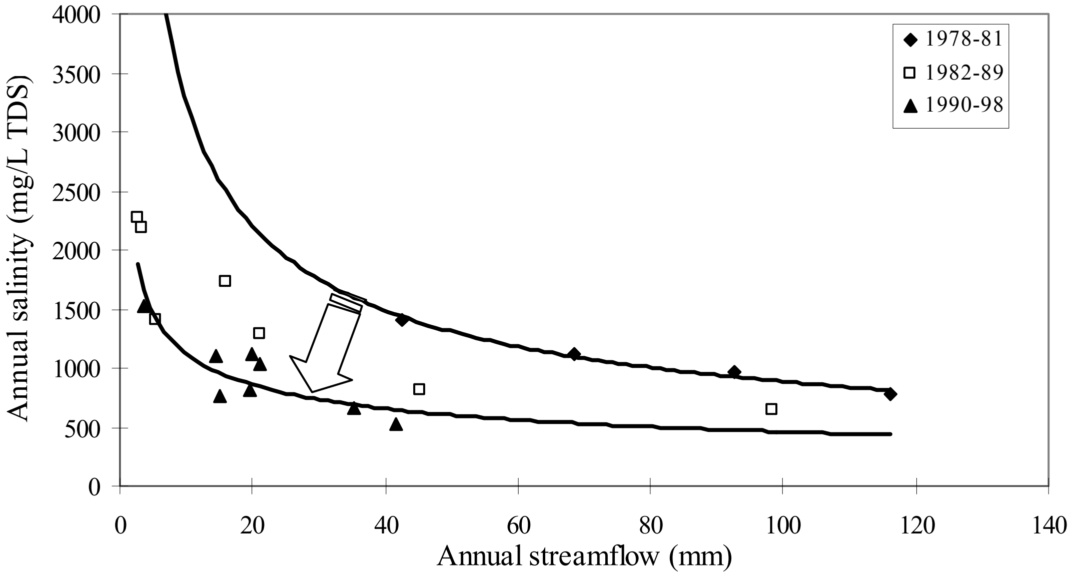

Figure 18. Relationship between annual streamflow and salinity in the Padbury Road catchment

6 Comparison of forest disturbances

A comparison of the impacts of various land use practices in the High Rainfall Zone of the jarrah forest of Western Australia is listed in Table 5. Except after clearing of native forest for agricultural development, water yields from all other treatments approach the pre-treatment levels within 10–12 years. However, the lag between treatment and the maximum increase in water yield depends on the method of treatment (Fig. 19). Clearing for agricultural development in the High Rainfall Zone had the longest lag (nine years). The lags of the other treatments are two years for logging and regeneration, four years for forest thinning, and seven years for bauxite mining and rehabilitation (Fig. 19).

A summary of the impacts of various land use practices on hydrological processes in all three rainfall zones is shown in Table 6. It shows that effects of forest thinning, operational logging and subsequent regeneration are temporary and cause less severe hydrological disturbances than clearing followed by agriculture. The streamflow responses to logging and subsequent regeneration are consistent with the results of other studies (Malmar 1992; Bren & Papworth 1991, Borg et al. 1988). A review by Bosch and Hewlett (1982) showed that reduction in forest cover typically increases streamflow. Cheng (1989) found that clear felling 34% of a catchment area produced clear and consistent streamflow increases. In the south-west of Western Australia, the annual streamflows of all treated catchments rose for 4 to 5 years and then began to fall as the vegetation regrew. The greatest increases were observed in the High Rainfall Zone and the smallest in the Low Rainfall Zone. The results are similar to those in earlier studies in Western Australia (Ruprecht & Stoneman 1993; Stoneman 1993; Ruprecht et al. 1991; Stoneman & Schofield 1989; Stokes & Batini 1985). It is clear that, 10–15 years after logging or forest thinning, the catchment returns to the pre-treatment conditions (Table 6). Studies in other parts of Australia have shown that the catchment yield may decline up to 50% depending upon the age of the regenerated forests (Jayasuria et al. 1993; Kuzera 1987). In Western Australia, there is insufficient data to quantify the long-term (> 20 years) effects of regenerated forest on streamflow and salinity.

Clearing followed by agriculture causes sustained increases in water yield, salt load and salinity. The salt load and salinity rises are more significant but take longer to be noticed as the annual rainfall decreases (Table 6). Work in the Recovery Catchments indicates that a new recharge–discharge equilibrium has been achieved after clearing controls some 25 years ago (Mauger et al. 2001, Mauger et al. 2003). In the areas with less than 600 mm annual rainfall, the salt-affected areas are still increasing and may continue to increase for another 50 years.

The studies to date have shown that logging and forest thinning in the High Rainfall Zone result in only temporary increases in stream salinity and stream sediments and that the annual stream salinity generally remained below 200 mg/L TDS. So, logging and forest thinning are not considered to adversely affect fresh water resources in this zone’s streams if the cleared areas can regenerate to native forest soon after logging. The potential for significant stream salinity rise after logging is highest in the Intermediate Rainfall Zone. In localised areas such as the March Road catchment (where groundwater was close to the surface and there are high soil salt stores) the annual stream salinity can be expected to exceed 500 mg/L TDS (Fig. 12). The risks and magnitude of increased salinity in the Intermediate Rainfall Zone are very variable and influenced by the degree of vegetation disturbance, the depth to groundwater and the soil salt storage. In the Low Rainfall Zone, the groundwater under native forest is generally deep enough not to mobilise salt in the soil profile. Any change in the annual recharge is small enough that logging can be undertaken without a significant rise in stream salinity.

Currently, bauxite mining is limited to the High Rainfall Zone. The effects of bauxite mining and rehabilitation on water yield were transient (Table 6). Once the rehabilitation was complete, water yields from all three study catchments returned to their pre-treatment levels (Fig. 16). To date, bauxite mining in the High Rainfall Zone has caused no significant impact on stream salinity. The lack of data means that the long-term (> 20 years) effects of rehabilitation on water yield and salinity are unknown.

Reforestation reduces streamflow, salinity and salt load and the effects on landscape hydrology are sustained (Bari & Boyd 1994; Bari 1998). After the logging and forest thinning, it takes about 10 years to achieve a new stability. Reforestation is one of the potential options for salinity management. Substantial plantings (more than 9000 ha) in the Collie River catchment have contributed to reducing stream salinity (Mauger et al. 2001).

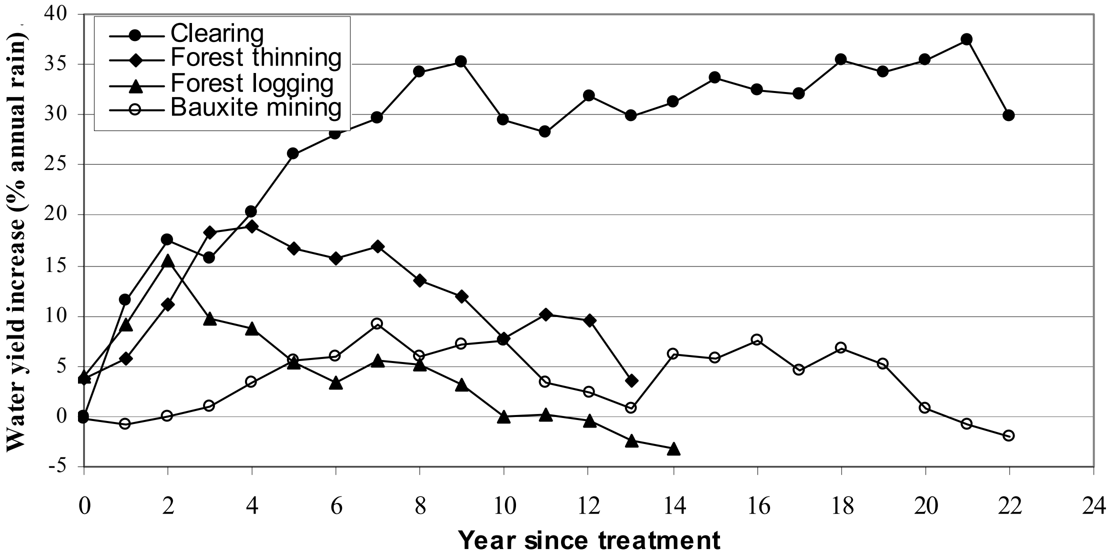

Figure 19. Comparison of water yield responses to land use practices in the High Rainfall Zone

Table 5. Comparison of water yield increases with land use practices in high rainfall areas of south-west Western Australia

Catchment	*Land use practice*	*Cover before (%)*	*Cover after (%)*	*Max increase in water yield (% rain)*	*Sustained increase in water yield?*
Wights	Clearing then agriculture	80	0	30	Yes
Lewin South	Logging & regeneration	90	20	14.5	No
Hansen	Forest thinning	60	14	20	No
Del Park	Mining and rehabilitation	N/A	N/A	9	No
More Seldom Seen	Mining and rehabilitation	N/A	N/A	21	No

Table 6. Summary of land use practices and expected impacts on flow regimes

Land use	*Expected water impacts*
Clearing for agriculture	Substantial increase in water yield and flood peaks Substantial salinity increases as mean annual rainfall decreases
Forest thinning and logging	Transient increase in water yield. The increase is likely to be greater with: – higher mean annual rainfall – more intense thinning or logging Transient, but small, increase in stream salinity. The higher salt risk areas are those where there is a combination of high soil salt storage and high groundwater levels. The current management responses to this risk include stream buffers and phased logging.
Bauxite mining and rehabilitation	Initial increase in water yield followed by decline back to pre-disturbance levels There are some uncertainties with long-term water yields possibly declining below pre-disturbance levels. Negligible salinity increases in higher rainfall areas.
Reforestation	Reduction in water yield — full effect likely to take 10–15 years. Reduction in salt load Impact on stream salinity can vary depending on level and location of reforestation.

7 Water yield issues in the management of the jarrah forest

The long-term (> 20 years) effects of increased water yield from areas subjected to forest thinning, logging or bauxite mining are still uncertain. There are two main reasons for this— limited data; and the relationship between vegetation cover, age, tree heights and evapotranspiration demand. About 20 years of data relevant to the effects of forest thinning, logging and bauxite mining in Western Australia have been collected. Detailed analyses of these data have demonstrated that the catchments return to their pre-treatment flow regimes within 10–15 years. Studies in other parts of Australia and overseas have shown that the longer-term (> 5 years) annual water yields may fall up to 50% below their pre-treatment volumes (O'Shaughnessy & Jayasuria 1991; Jayasuria, et al. 1993; Kuzera 1987). Measurements of the crown cover and basal area have shown that, after logging or forest thinning, the jarrah forests return close to their pre-treatment vegetation cover within 10–15 years (Borg & Stoneman 1988; Stoneman et al. 1989; Bari & Boyd 1993; Robinson et al. 1997). If the vegetation cover exceeds the pre-treatment cover and the water yield is reduced at the same time, appropriate management strategies (e.g. thinning) should be considered to enhance water yield.

Climate variability is another factor directly affecting the catchment-scale water yield. Annual rainfall, particularly during the winter months, has declined by more than 10% in the south-west of Western Australia (IOCI 2002). As a consequence of this lower rainfall, the total inflow to the reservoirs has declined by 41% during the last 25 years (Fig. 5). If the annual rainfall were not lower, then the effects of forest treatment on water yields would have been different. It could have taken longer to return to the pre-treatment levels and there would have been higher yields mainly due to higher recharge to the groundwater system, and reduced transpiration relative to rainfall.

Most of the water supply reservoirs for the city of Perth are located within the jarrah forest of Western Australia (Fig. 1, Table 7) so forest thinning to enhance water yield has a high potential to increase streamflow to these reservoirs. In fact, it had been reported that selective forest thinning within the water supply catchments has the potential to increase inflow to the reservoirs by as much as 50% (Stoneman & Schofield 1989). The cost of producing the extra water by thinning is estimated to be 23 cents per kilolitre (Colin Terry, *pers. comm.*), substantially less than any other alternative water resource development (Stoneman & Schofield 1989).

Table 7. Water yield from major surface water supply catchments for Perth

Water supply Catchment	*Catchment area (km^2)*	*Rainfall Zone*	*Mean annual water yield* (mm)	(% rain)
Mundaring	1482	I–L	26	3
Canning	727	I–L	72	7.5
Wungong	130	H	197	15
Serpentine	660	H–I	110	10
North Dandalup	153	H	183	14
South Dandalup	315	H–I	97	9
Total	**3 467**		**71**	**7**

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