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Hydrological response to bauxite mining and rehabilitation in the jarrah forest in south west Australia

Andrew H. Grigg[a,b]

[a] Environmental Department, Alcoa of Australia, Australia
[b] Huntly Mine, PO Box 172, Pinjarra, WA, 6208, Australia

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ABSTRACT

Study region: Jarrah forest in south west Australia.

Study focus: The hydrological response to bauxite mining in the jarrah forest could differ from other land uses such as timber harvesting or clearing for agriculture, since mining involves excavation of the upper regolith in addition to changes in forest cover due to clearing and subsequent rehabilitation. Three catchments, one subject to mining, a second subject to an intensive forest thinning treatment and an untreated control were monitored for streamflow, rainfall, groundwater and leaf area index over a 36-year period.

New hydrological insights for the region: Mining caused a peak streamflow response of 225 mm or 18% of rainfall, before returning to pre-disturbance levels 11 years after mining commenced. Streamflow changes were closely associated with changes in a groundwater discharge area in the valley floor. Changes in groundwater level, in turn, were related to rainfall and leaf area index, and these effects did not differ between mine rehabilitation and unmined catchment areas. The streamflow response to mining could not be distinguished from the intensive thinning treatment in this study, or from clearfelling or clearing for agriculture reported elsewhere in the jarrah forest. The results indicate that shallow subsurface flow processes, considered to dominate streamflow generation in jarrah forest catchments, do not extend beyond the valley floor and immediately adjacent slopes which were not disturbed by mining.

1. Introduction

The deep highly weathered lateritic profiles that support jarrah (*Eucalyptus marginata*) forests in south-west Western Australia are capable of storing a large proportion of annual rainfall (Schofield et al., 1989). The store of soil water is exploited by the extensive rooting system of jarrah to depths of 40 m or more (Dell et al., 1983) and evapotranspiration forms the major loss component of the jarrah forest water balance, estimated in catchment studies to exceed 90% of annual rainfall (Ruprecht and Stoneman, 1993). Hence, manipulation of forest cover has long been proposed as one option to influence catchment yields (Stoneman and Schofield, 1989) and numerous studies have been undertaken to determine catchment responses to forest harvesting activities (Ruprecht et al., 1991; Stoneman, 1993; Bari et al., 1996; Robinson et al., 1997; Kinal and Stoneman, 2011). In reviewing the impacts of land use practices in 27 catchment studies across the south-west of Western Australia, Bari and Ruprecht (2003) reported that clearing for agriculture led to permanent increases in yield of about 30% of annual rainfall in high rainfall (> 1100 mm) areas. Forest thinning in higher rainfall areas resulted in maximum streamflow increases of 8–18% of rainfall, depending on the degree of treatment. Streamflows returned to pre-treatment level after 12–15 years, matching vegetation recovery, or longer if regeneration is limited.

E-mail address: andrew.grigg@alcoa.com.au.

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The lateritic profiles also support extensive but discontinuous and shallow (3–5m) surface deposits of bauxite, which have been mined since the 1960′s (Hickman et al., 1992). Expansion of mining in the 1970′s raised concerns over its effects on the hydrology of the jarrah forest (Steering Committee, 1978) and a small number of empirical catchment studies investigating the effect of mining have been reported. Ruprecht and Stoneman (1993) found that mining of 16% of Del Park catchment in 1975–79 resulted in a peak yield increase of 8% of rainfall, followed by a return to pre-mine levels 12 years after the commencement of mining. Bari and Ruprecht (2003) reported larger peak responses in the Seldom Seen and More Seldom Seen catchments of 23% and 21% of rainfall, respectively, noting a good correlation between the increase in streamflow and the proportion of the catchment cleared for mining but not yet rehabilitated. Croton and Reed (2007) found peak increases of 200–250 mm/year, representing responses of 14–17% of rainfall, in a further two mined catchments. In all cases, a consistent pattern of an initial increase in flow followed by a return to, or below, pre-mining levels was observed. These patterns show similarities to the responses observed for other land use practices, however, short (1–3 year) pre-mining calibration periods or difficulties with suitable controls detracted from some of these studies and none went beyond a consideration of annual flow responses. Furthermore, Croton et al. (2005) claimed that a higher water use in young mine rehabilitation was necessary to obtain an acceptable match to streamflow in modelling studies. There remains, therefore, a need to understand in greater detail the effects of bauxite mining on hydrological processes than has been reported to date.

Concurrent with the effects of land use practices on streamflows in the jarrah forest has been the effect of a drying climate. The south-west of Australia has experienced a 15–20% decline in annual rainfall since the 1970′s and a growing number of once perennial streams in the higher rainfall parts of the forest are now seasonal (Petrone et al., 2010). Streamflow decline is observed as a step change in response to the occurrence of years of very low rainfall, reflecting a strong correlation between runoff as a proportion of rainfall and groundwater storage (Hughes et al., 2012). Catchment groundwater storage increases when rainfall exceeds a certain threshold but decreases in years when rainfall is below the threshold. Kinal and Stoneman (2012) reported a particularly dramatic drop in streamflow when groundwater declined below or 'disconnected' from the valley floor, highlighting an 'amplifying' role of groundwater in streamflow generation. When groundwater levels are well below the valley floor, even intensive forest thinning within a catchment can have no effect on streamflows (Kinal and Stoneman, 2011).

The aims of this study were to determine the hydrological response to bauxite mining and subsequent rehabilitation in the jarrah forest, and to compare the response to mining with the response to other land use practices. The study utilised three small jarrah forest headwater catchments over a combined experimental period of 36 years. One catchment experienced a 5-year period of mining and associated rehabilitation, a second was subject to an intensive thinning treatment, and a third acted as an untreated control. Comparisons were made between the mined catchment and the untreated control to determine the effects of mining independent of changes due to climate, while the intensively thinned catchment provided a comparison between a mining disturbance and an alternative land use practice that reduced catchment forest cover to an extent similar to the mined catchment but without excavation of the upper regolith. Detailed measurements of rainfall, groundwater, streamflow and changes in forest leaf area index (LAI), a key determinant of vegetation water use (Waring, 1983), were collected and are reported here.

2. Materials and methods

2.1. Geomorphology, climate and bauxite mining in the jarrah forest

The northern jarrah forest region of Western Australia occurs on the Darling Plateau, an elevated undulating landform developed predominantly on coarse-grained granites and granitic gneisses (Churchward and Dimmock, 1989). The basement rock has been weathered *in situ* to form deep (> 30m) lateritic profiles, the upper parts of which are enriched in sesquioxides of iron and aluminium. The surface horizon consists typically of gravels, sands and loams including a discontinuous indurated layer or duricrust, mostly in mid- to upper-slope positions, merging with the underlying mottled and pallid clay zones. The sandy gravels of the upper slopes become finer downslope, forming deep sands adjacent to the valley floor which in turn are typically dominated by loams and clay loams (Churchward and Dimmock, 1989). Root channels of lower bulk density extending vertically through fissures and discontinuities in the indurated layer and deep into the mottled and pallid clay zones (Dell et al., 1983) are a feature of the lateritic profiles, forming preferred flow paths for infiltrated rainfall and permitting rapid recharge of permanent groundwaters (Johnston, 1987).

The climate of the region is Mediterranean with winter-dominant rainfall (May to October) and a summer drought. Rainfall is greatest on the western margin of the jarrah forest and declines with distance inland. Historical annual average rainfall ranged from 1300 to 600 mm (Gentilli, 1989), however, the region has experienced a 15–20% rainfall reduction since the 1970′s and drought years are now more frequent (Petrone et al., 2010).

The alumina-rich duricrust and mottled zone materials constitute the bauxite ore removed by mining (Hickman et al., 1992). Alcoa of Australia (Alcoa) has been mining for bauxite in the northern jarrah forest since 1963 and presently clears and rehabilitates approximately 550 ha annually (Koch, 2007a). Alcoa's operations comprise a mosaic of shallow pits averaging 4 m in depth and around 20 ha in size distributed across a mining region and linked to a centrally-located crusher by a radiating network of haulroads. A detailed description of the mining process and rehabilitation prescriptions is provided in Koch (2007a). Briefly, the process involves harvesting and clearing of the native forest, stripping of topsoil and subsoil layers to expose any lateritic duricrust layer present, followed by blasting and extraction of the duricrust and underlying friable bauxite. Once ore has been removed, the pit is landscaped to form an undulating terrain while ensuring that surface water does not discharge from the pit into adjacent unmined areas. Ripping using a winged tine to an approximate depth of 1.5 m is undertaken to relieve compaction of the pit floor, the subsoil and topsoil are returned, and the surface ripped for a second time to approximately 0.8 m depth along the contour. This aids infiltration, reduces the

Table 1
Selected characteristics of the three catchments.

Characteristic	Lewis	Hansen	Bates
Treatment	Mining	Intensive thinning	Control
Area (ha)	186	76	230
Weir elevation (m AHD)	277	256	255
Max. slope (°)	7	11	10
Max. relief (m)	80	70	70
Previous silvicultural treatments	Selective logging 1940s–1950s	Selective logging 1940s–1950s	Logging 1920s, 1980–83 (60% of area), burns 1984 and 2001
Flow record	1978–2013	1978–1998	1989–2013

potential for erosion and provides a tilled seedbed for subsequent seed application. Seed mixtures of 73–113 tree and understorey species are broadcast onto the ripped ground in the summer and autumn, and additional nursery-raised seedlings are planted by hand in the first winter. No further trafficking of the rehabilitated surface takes place and the vegetation is allowed to develop. The objective of rehabilitation is to restore a functioning jarrah forest ecosystem containing the dominant overstorey species of jarrah and marri (*Corymbia callophylla*) and a diverse mix of understorey plants (Koch and Samsa, 2007; Koch, 2007b).

2.2. Experimental catchments

Three catchments were examined in this study (Table 1). Lewis was subjected to standard mining operations and Bates acted as an untreated control. Hansen catchment had been intensively thinned in an earlier experiment and results have been previously reported (Ruprecht et al., 1991; Robinson et al., 1997). This catchment was used in the present study as a comparison between mining and non-mining disturbances with similar reductions in catchment forest cover, with analysis in the present study extending the previously reported results to include an examination of the dynamics of the groundwater discharge area and changes in catchment leaf area index. Lewis has a long streamflow record commencing in 1978, acting initially as a control for comparison with Hansen catchment (Ruprecht et al., 1991), before being mined in the late 1990′s. Bates catchment was established in the late 1980′s as a new control for comparison with Lewis in the present study.

All three catchments are located in the higher rainfall area of the northern jarrah forest approximately 100 km SE of Perth (32°38′S, 116°06′E), 10 km to the north of the town of Dwellingup and within a distance of 2 km of each other (Fig. 1). The long-term average rainfall for the area containing the catchments was approximately 1300 mm, while the long-term mean annual pan evaporation was 1600 mm (Ruprecht et al., 1991; Robinson et al., 1997). Lewis catchment supported an open forest dominated by jarrah and marri on the mid and upper slopes with bullich (*Eucalyptus megacarpa*) and swamp heath in the riparian zone primarily at the junctions of the two secondary stream branches. A mid-storey of bull banksia (*Banksia grandis)* was variably present along with grasstrees (*Xanthorrhoea preissii)* and a range of schlerophyllous shrubs and ground layer species (Bell and Heddle, 1989). Mining disturbance in Lewis commenced with clearing for haulroads and pit development in October 1996, after the main winter rainfalls. Additional areas were progressively cleared and mined through to February 2000 (Fig. 1). Rehabilitation of mined pits commenced in 1998 as mined out areas became available, continued in 1999 and all areas were completed by May 2001. There were no areas rehabilitated in the year 2000. A total of 51% of the catchment area was disturbed, with mine pits located on mid slope and upper slope positions. None of the riparian areas were disturbed (Fig. 1). In 2003, a fuel-reduction burn affected unmined portions of the catchment but none of the areas of young rehabilitation.

Bates catchment contained jarrah and marri forest regenerating from harvest activities in the 1920′s and most recently in the early 1980′s (Table 1), with blackbutt (*Eucalyptus patens*) in the valley floor. A mid-storey was largely absent while zamia (*Macrozamia riedlei)* and *Trymalium ledifolium* were common in the ground layer. Species indicative of wetter conditions including *Taxandria linearifolia*, *Astartea fascicularis* and *Melaleuca preissiana* were prevalent along the main watercourse. A post-harvest burn occurred in 1984 (prior to the commencement of stream records) and a fuel-reduction burn across the catchment was undertaken in 2001.

Hansen catchment was described in detail by Ruprecht et al. (1991). Briefly, the catchment was initially covered by open forest of jarrah and marri with a patchy sclerophyll understorey generally less than 1 m tall. A swamp occupied the lower central part of the catchment. In the summer of 1985–1986, a uniform and intensive thinning treatment was applied across the catchment, excluding the swamp and a 50 m buffer, reducing tree basal area from 27–35 m^2/ha down to 7 m^2/ha (Ruprecht et al., 1991; Robinson et al., 1997).

2.3. Instrumentation and measurement

Hydrological measurements in the three catchments encompassed streamflow, rainfall and groundwater. Streamflow was measured using sharp-crested V-notch weirs (90° in Lewis, 90° in Hansen from 1978 to 1994 and 60° from 1994 to 1998, 120° in Bates) with deep (2 m) cut-off walls. Rainfall was measured by a pluviometer located in a forest clearing either at the outlet (Lewis, Hansen) or higher in the catchment (Bates) (Fig. 1). Streamflow and rainfall were recorded in Lewis and Hansen from late 1977 while streamflow recording in Bates commenced in mid 1988. Monitoring in Hansen was discontinued in early 1999, while monitoring continued in Lewis and Bates through to the end of 2013. Rainfall measurement in Bates commenced in May 1992 and annual rainfall for the period 1989–1992 was estimated from a regression with annual rainfall in Lewis. The earliest full year common to the mined

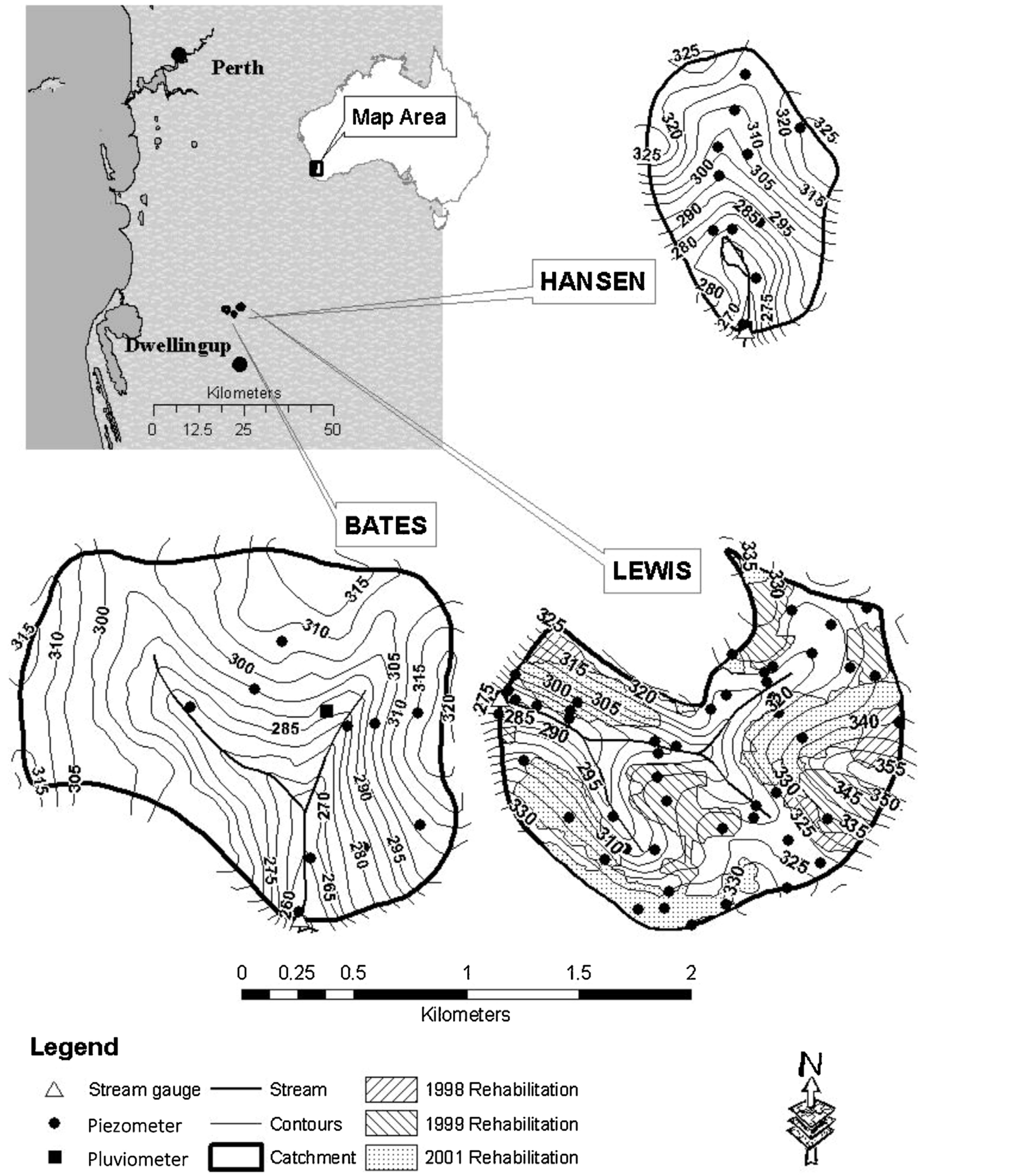

Fig. 1. Location of the three experimental catchments in the study region and topography of each catchment showing location of mining and rehabilitated areas in Lewis, stream gauging stations, pluviometers and groundwater monitoring piezometers.

(Lewis) and control (Bates) catchments for which streamflow data were available was 1989.

Groundwater monitoring commenced in Lewis in late 1988. Measurement of depth to the permanent groundwater was typically monthly. Small gaps in measurements were infilled by linear interpolation, however, where more than half the annual cycle was missing, or the piezometer was dry for more than half the year, the record was not used in subsequent calculations. The number of deep piezometers available in each individual year varied between 7 and 41 over the study period, and a total of 55 over the study period. The number increased from 26 to 41 in 2000–2001 as piezometers were installed in previously mined and rehabilitated areas. For the period 1989–1992 when only seven piezometers were available, groundwater depths for an additional 12 locations were estimated as follows. For the period 1993, when new piezometers were constructed, to the end of 1996 when clearing for mining commenced, correlations between the seven original piezometers and the piezometers constructed in 1993 were generated. The best regression models, with correlation coefficients ranging from 0.88 to 0.98, were then applied using measurements from the original seven piezometers for the earlier period. In Bates catchment, records from a total of 12 deep piezometers were available commencing in late 1988, although two of these became dry within the first five years due to falling groundwater levels. A third piezometer became permanently dry from 2010. A total of 13 deep piezometers were established across Hansen in April 1984 (Ruprecht et al., 1991). For this study, a total of 11 piezometers with approximately monthly readings from April 1985 to early 1999 were used.

An annual time series of spatially averaged catchment LAI was calculated (i) for each catchment, (ii) for individual areas of rehabilitation within Lewis catchment excluding pixels that overlapped rehabilitation boundaries, and (iii) for defined zones associated with individual piezometers in Lewis and Bates catchments (see below). Data were derived from mapping of combined canopy and understorey leaf area index generated for the northern jarrah forest using a series of standardised and calibrated Landsat imagery developed and described by Macfarlane et al. (2017).

2.4. Data analysis

2.4.1. Streamflow response

A linear regression of annual streamflow was established between Lewis and Bates catchments for the pre-mining period. The regression was then applied to the flow data for Bates in the post-mining period to predict flow in Lewis as if there had been no mining.

2.4.2. Groundwater

The response of the deep groundwater to mining in Lewis catchment was estimated as the difference in annual average depth to water compared to a suitable control piezometer in Bates catchment. For piezometers in unmined mid- to upper-slope locations, an average of three control piezometers in Bates was used to reduce the potential for error. One of these bores became dry after 2010 and an average of two bores was used for the final three years of record. For lower slope and streamzone positions, only a single control piezometer was used due to limited availability of piezometers in Bates in comparable locations. In addition, there were too few water level readings in Bates during 1998 to calculate an average depth to water, and a linearly interpolated value from adjacent years was used instead. Differences between each piezometer in Lewis and respective controls in Bates were then normalised by substracting the average difference for all available pre-mining years up to and including 1996. For piezometers established in rehabilitated locations after mining where there were no pre-mining level data, differences between Lewis and Bates were normalised by subtracting the difference recorded in the year that a peak was either observed or, in a small number of cases, estimated.

An analysis of year-on-year changes in annual average groundwater level at the piezometer scale in Lewis and Bates catchments was undertaken to investigate whether groundwater levels responded differently between mined and unmined locations. A generalized linear modelling approach was implemented in Minitab 17 (Minitab Inc., Pennsylvania, USA) using the average of the current and preceding years' rainfall and LAI as main effects. LAI was estimated as the average for a 100 m wide strip extending from the individual piezometer perpendicular to the contour upslope to the closest ridgeline. For piezometers close to the catchment divide, a circle of radius 50 m was used. Piezometers located in valley floor or streamzone positions were excluded from the analysis, giving a total final dataset of 497 records. The location of the piezometer in either Bates or Lewis, and for Lewis in either an unmined or mined and rehabilitated part of the catchment, was included as a random factor in the model. Subsequently, interaction terms and main effects were dropped in a backwards stepwise elimination with an $\alpha = 0.05$ to remove until only significant terms remained in the model (Sokal and Rohlf, 1995).

Streamflow responses to catchment disturbance in the jarrah forest have been closely linked to the size of a groundwater discharge area within the valley floor (Ruprecht and Schofield, 1989; Silberstein et al., 2003). Therefore, estimates of the extent of such zones were made in the present study for each year of available groundwater records using the following approaches. In Bates and Hansen, an annual average depth to water below the ground surface was calculated for each piezometer using all available measurements for the year. There were insufficient piezometers in either catchment to generate a reliable interpolated piezometric surface from these data alone. Instead, mean annual depth to groundwater for each piezometer was regressed against values of UPNESS at each piezometer. UPNESS is a topographic index variable within the model FLAG (Roberts et al., 1997) that describes a position within the landscape in terms of the size of the contributing area, calculated from the number of gridcells connected by a monotonic continuous uphill path. An asymptotic regression model of the form $y = a - b*exp(-c*x)$ was fitted to the annual depth to water and UPNESS data for each catchment, and maps of annual depth to groundwater were generated by applying the predicted depth of water to a 50 m grid of UPNESS values constructed over each catchment. Maps were generated within Surfer 8.0®, using the default linear variogram and point kriging options. In a small number of years, regressions could not be performed due to insufficient piezometer readings, and regression coefficients were estimated by interpolating between adjacent years. In Lewis, a different approach was required as forest cover across the catchment was not uniform as a consequence of mining, and groundwater depth could not reliably be estimated from UPNESS alone. In this catchment, the larger number of piezometers enabled the generation of contour maps of depth to groundwater directly from the piezometric dataset. For all catchments, the extent of groundwater discharge areas was delineated where mean annual groundwater level was within 2 m of the topographic surface, which is conservatively indicative of groundwater 'connection' in the riparian zone (Hughes et al., 2012). In these catchments with intact riparian vegetation, groundwater rarely intersects the surface all year round (Kinal and Stoneman, 2012).

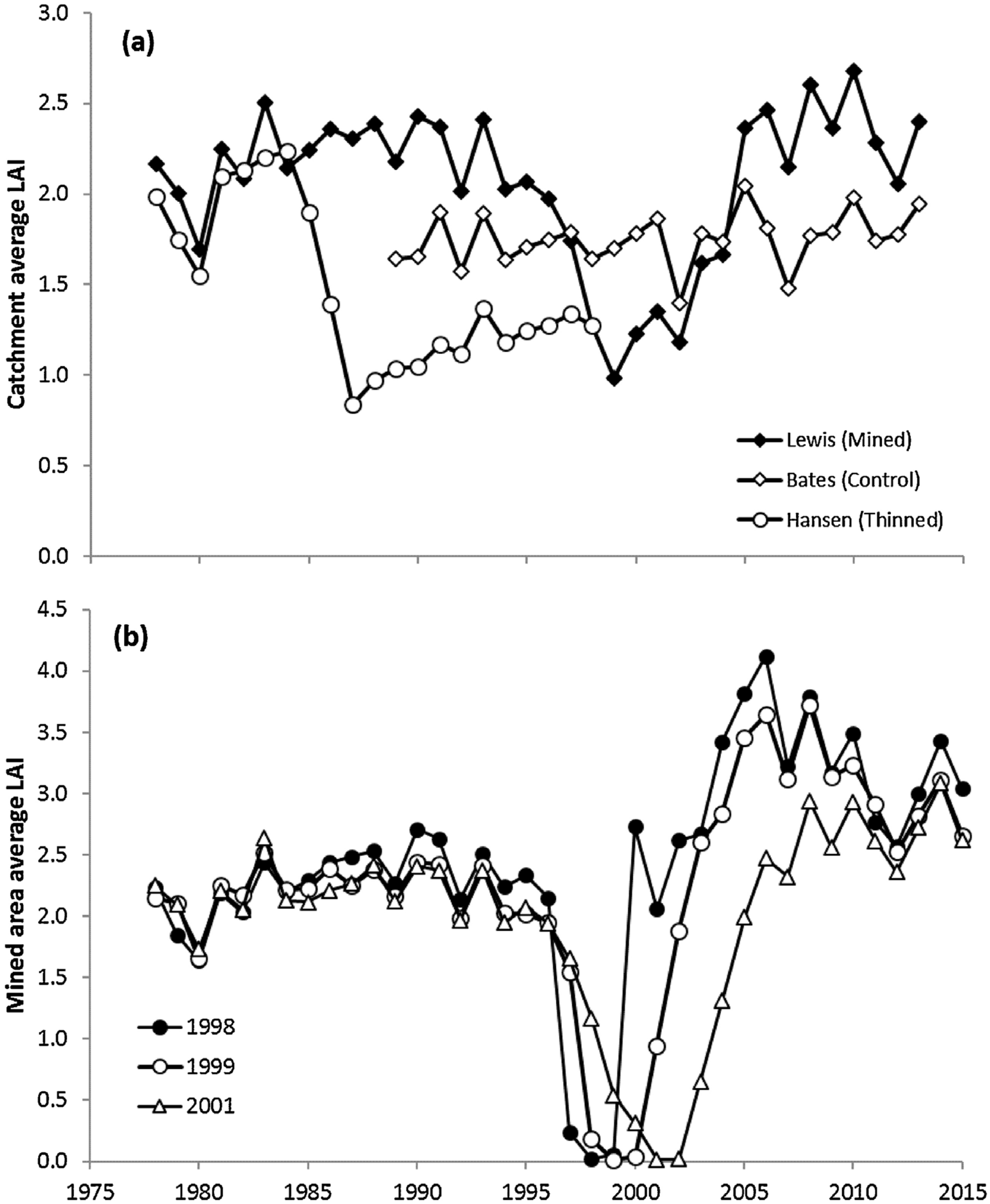

Fig. 2. Changes in (a) catchment average LAI in the three catchments over the periods monitored for streamflow, and (b) mean LAI within areas of Lewis catchment that were mined and subsequently rehabilitated in three different years.

3. Results

3.1. Rainfall

In Lewis catchment, annual average rainfall for the pre-mining period 1989–1996 was 1209 mm, 5% above the long-term annual average (1155 mm), but in the post-mining period 1997–2013 was 1075 mm or 7% below the long-term annual average. Substantially drier years were observed in 2001 (771 mm), 2006 (841 mm) and 2010 (605 mm), representing reductions of 33%, 27% and 48%, respectively, compared to the long-term average. Similar patterns were evident for Bates catchment which had an average annual rainfall of 1192 mm in the period 1989–1996, and 1049 mm in the period 1997–2013. Reduced annual rainfall in the post-mining period was associated with lower rainfall in the months of May, June and July. In Hansen, annual average rainfall for the pre-treatment period 1978–1985 was 1214 mm, and 1217 mm for the post-treatment period 1986–1998.

3.2. Changes in catchment and mined area LAI

Catchment average LAI in Lewis was more than halved from approximately 2.2 prior to mining to a minimum of 1.0 in 1999 (Fig. 2a) when most clearing had been completed and areas of young rehabilitation supported minimal vegetation cover. Recovery in LAI was rapid from 2002 when all mining and rehabilitation activities had been completed, and stabilised at a catchment average LAI of 2.4 from approximately 2005. Catchment average LAI in Bates showed a slight upward trend throughout the period of records, rising from 1.6 to approximately 1.9 between 1989 and 2013 (Fig. 2a). The marked reduction in catchment average LAI in Hansen to a minimum LAI of 0.8 in 1987 due to the intense thinning in 1985–1986 is also evident (Fig. 2a). The magnitude of the reduction in Hansen is comparable to the peak reduction in LAI in Lewis due to mining, while in contrast to Lewis, recovery was much slower.

In Lewis, all areas that were subject to mining supported forest with an average LAI of 2.2 over the period 1978–1996 (Fig. 2b). Recovery in LAI was rapid in the second year after rehabilitation establishment, attaining pre-mine levels within approximately six years, and attaining an equilibrium average LAI of 3.0 shortly thereafter (Fig. 2b). Routine monitoring undertaken in all rehabilitated pits one year after establishment indicated that pits in Lewis rehabilitated in 1998 and 1999 contained an average combined stocking of jarrah and marri of 3200 trees/ha (Alcoa, unpublished data), and a lower average of 2000 trees/ha for 2001 rehabilitation. Understorey densities were also high, averaging close to 5 plants/m^2 for 1998 and 1999 rehabilitation and 2.5 plants/m^2 in 2001 rehabilitation. Results of a single plot measurement in 2008 (unpublished data) of rehabilitation established in 1999 contained 2100 trees/ha with a basal area (over bark) of 28 m^2/ha (LAI 1.5) and a tall dense understorey dominated by *Bossiea aquifolium* with a notably high LAI of 1.4.

3.3. Groundwater

Groundwater hydrographs for mid-slope upper catchment, mid-catchment valley and catchment outlet locations in unmined portions of Lewis, and comparative locations in Bates, are shown in Fig. 3(a–c). While there is a break in the record at the upper location in Lewis due to access restrictions, a rise in groundwater due to mining immediately upslope from 1997 and a minimum depth to water in or before 2000 are clearly visible (Fig. 3a). Over the same period, there is a relatively steady decline in groundwater levels in Bates, representing a total decline of 8.92 m at an average rate of 0.42 m/year. In the mid-catchment valley locations, groundwater depth minima are apparent in 2000, 2002 and 2003, although not in the dry year of 2001 (Fig. 3b). These years are also associated with a reduction in the seasonal amplitude of the hydrograph. Step declines of similar magnitude in Lewis and the control catchment Bates coinciding with drought years in 2006 and particularly 2010 are evident. At the catchment outlet, seasonal fluctuations and the effects of the dry years in 2001 and 2006 are more muted than higher in the catchment (Fig. 3c). However, the effect of the 2010 drought in both Lewis and Bates is clearly recognisable, characterised by significant declines in average groundwater levels and a marked increase in the amplitude of the seasonal cycle thereafter.

Changes in groundwater level in the catchment before, during and after the period of mining are shown relative to equivalent unmined groundwater levels in Fig. 4(a–c). All time traces showed a consistent pattern, rising rapidly after clearing to a peak before returning to pre-mine levels, or slightly below pre-mine levels. In mid-slope locations, peak rises of 2.5–5 m occurred within the period 1999–2003 (Fig. 4a), while slightly higher peaks of 2.8–5.5 m were observed in lower slope and valley edge locations, typically around 2003 (Fig. 4b). Peaks were much smaller in streamzone locations or near the catchment outlet, in the range 0.5–1.5 m (Fig. 4c). In all three sets of locations, groundwater levels in nearly all cases had returned to, or declined slightly below, pre-mining levels relative to the unmined state by 2008, approximately 11 years after initial mining disturbance. For piezometers established in mine pits after final rehabilitation, peaks in groundwater levels were typically observed in 2002–3 with subsequent declines of 2.5–5.5 m (Fig. 4d). While levels in a number of piezometers appeared to equilibrate from about 2008, several others continued a declining trend relative to the control.

Both rainfall and LAI were highly significant ($P < 0.001$) in predicting the year-on-year change in mean annual depth to groundwater, with the model explaining 53% of the variation. Piezometer location in Bates or Lewis, or in mined or unmined parts of either catchment, had no significant effect, and there were no significant interactions. The effect of rainfall was particularly strong (Fig. 5): two-year average rainfalls in excess of 1200 mm were almost always associated with reductions in the depth to groundwater (ie. groundwater rise), while conversely, two-year average rainfalls less than 1100 mm resulted in groundwater falls, except where LAI was low. More generally, LAI was inversely related with change in depth to groundwater.

In Bates, the maximum extent of the groundwater discharge area occurred in 1992 followed by an almost unbroken decline throughout the rest of the record (Fig. 6a). Step declines were apparent around the dry years 2001 and 2010, and after the latter only a very small groundwater discharge area remained at the catchment outlet. In Lewis, a peak in the size of the groundwater discharge area was also evident in 1992, followed by an expansion after mining entry in 1997 to a maximum extent of 8% of the catchment area in 2000 (Fig. 6b). The discharge area contracted in 2001 and further again in 2002, recovered to some extent in 2003 but declined thereafter. There was no groundwater discharge area in the catchment on an average annual basis from 2010 onwards, although transient connection in the lowest parts of the catchment was observed for several weeks in the winter and spring of 2010–2013. In Hansen, the groundwater discharge area expanded rapidly in 1987 following thinning treatment in 1985–86, peaking at an average 12.4% in the period 1989–1992 and then declining to the end of records in 1998 (Fig. 6c).

3.4. Streamflow

Annual streamflow and runoff coefficient in Bates catchment exhibited a declining trend throughout the study period from a peak

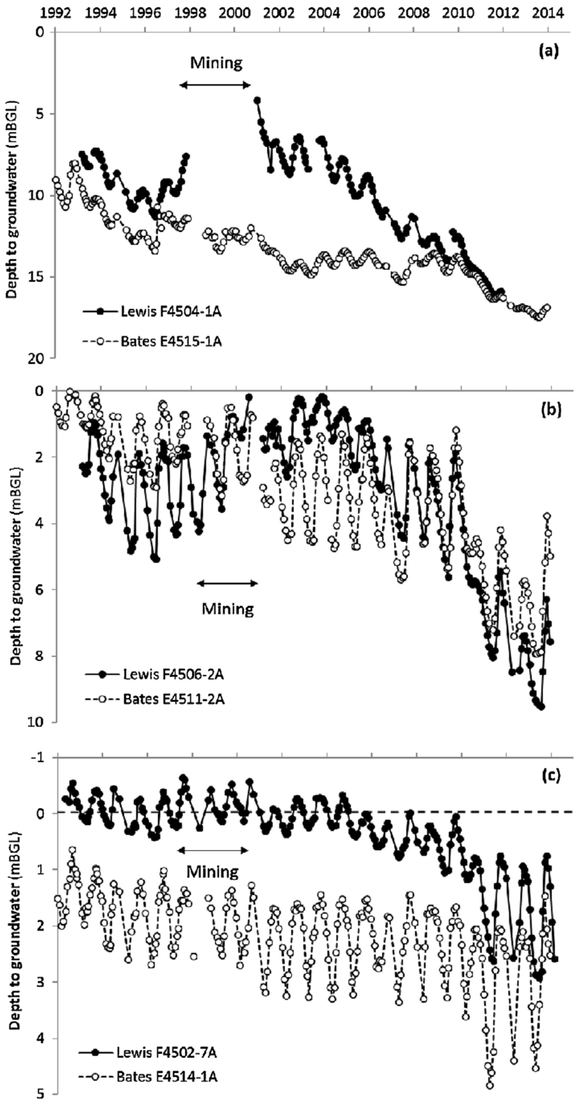

Fig. 3. Groundwater level hydrographs for selected piezometers in (a) upper catchment mid-slope, (b) mid-catchment valley and (c) catchment outlet locations in Lewis (mined) and comparable piezometers in Bates (Control). The periods of mining from clearing to final rehabilitation in areas upslope of each piezometer in Lewis are indicated.

of 325 mm or 27% of rainfall as runoff in 1992 to a low of 28.8 mm or 2.6% of rainfall in 2011 (Fig. 6a) following the record dry year of 2010. This catchment changed from perennial to seasonal flow after 2010, with flows in the last three years 2011–2013 limited to the months of June through December. In Lewis catchment, the runoff coefficient was initially low before a peak following well above average rainfall in 1991, followed by a higher peak of 307 mm or 19% of rainfall in 2000 (Fig. 6b) a year after the lowest average catchment LAI (Fig. 4a). Flow fell sharply in 2001 corresponding with an exceptionally dry year, recovered slightly in the subsequent two years but then declined for the remainder of the study period (Fig. 6b). Zero-flow days during the study period were observed in autumn 2008 (17 days) and increased in frequency thereafter (2009, 110 days; 2010, 209 days; 2011, 220 days; 2012 298 days; 2013, 285 days). Streamflow in Hansen increased after thinning to a peak of 438 mm or 33% of rainfall in 1992 (Fig. 6c). Annual flows declined in each subsequent year to the end of record in 1998 except for the relatively wetter year of 1996. Consistent across all three catchments, there is a close match throughout the periods of record between the runoff coefficient and the size of the groundwater discharge area.

Annual streamflows in Lewis and Bates catchments were closely related during the pre-mining period 1989–1996, with the exception of the years 1989 and 1990 which have been excluded in the fitted regression (Fig. 7). In the two years prior to 1988, Lewis catchment exhibited runoff coefficients of approximately 0.04 and days of zero-flow. Additionally, inspection of the hydrographs for daily flow revealed that during 1989 and 1990, streamflow in Lewis did not respond to rain events early in the winter rainfall season, remaining low until July. Later in the season, both catchments responded in a similar pattern (data not shown). In Fig. 7, annual flows from 1997 to 2007 in Lewis lie above the regression line, while annual flows from 2008 to 2013 (except 2009) form a continuation of

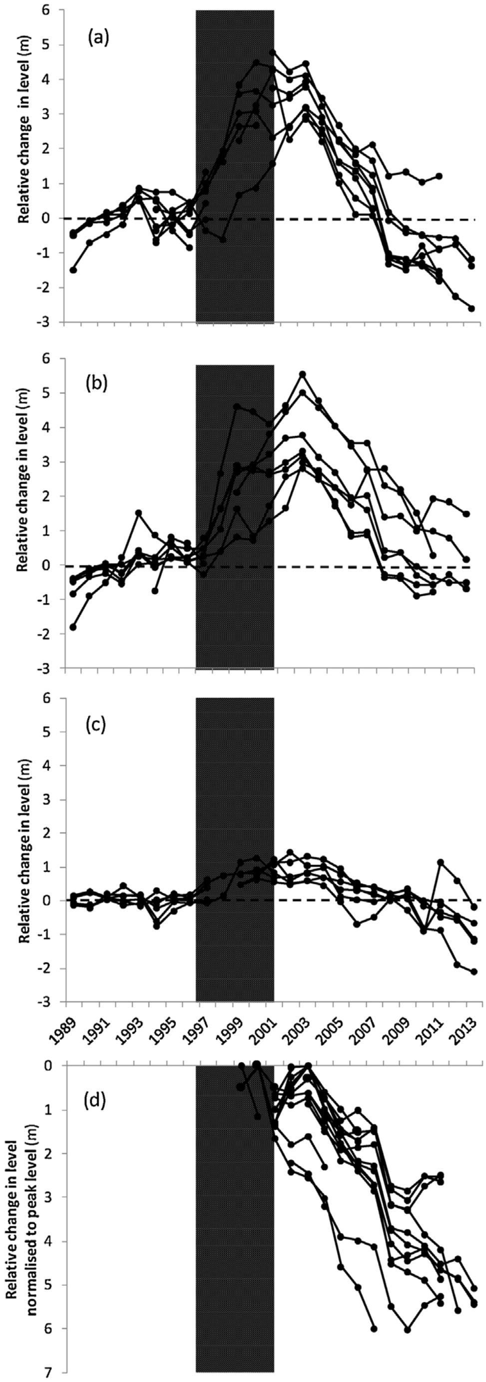

Fig. 4. Mean annual groundwater level relative to an unmined control for piezometers in Lewis in (a) mid-slope, (b) lower slope and valley edge, (c) streamzone locations. Note that in (d) rehabilitated areas, groundwater levels have been normalised to the observed or estimated minimum depth to water in each piezometer, since no pre-mine record exists. The shaded interval indicates the period of mining from clearing to final rehabilitation.

the pre-mining relationship, with both catchments displaying lower flows than during the pre-mining period.

The fitted regression shown in Fig. 7 was given by (Eq. (1)):s

$$Q_L = 0.74 * Q_B - 31 \ (R^2 = 0.99, P < 0.01) \tag{1}$$

where Q_L is the Lewis catchment streamflow (mm) and Q_B is annual flow in Bates (mm). This linear relationship predicts negative flows in Lewis catchment for annual flows in Bates less than 42 mm which occurred from 2010 onwards, therefore estimates of flow response in Lewis using Eq. (1) (Table 2) will be an overestimate. The true relationship between the two catchments at these low

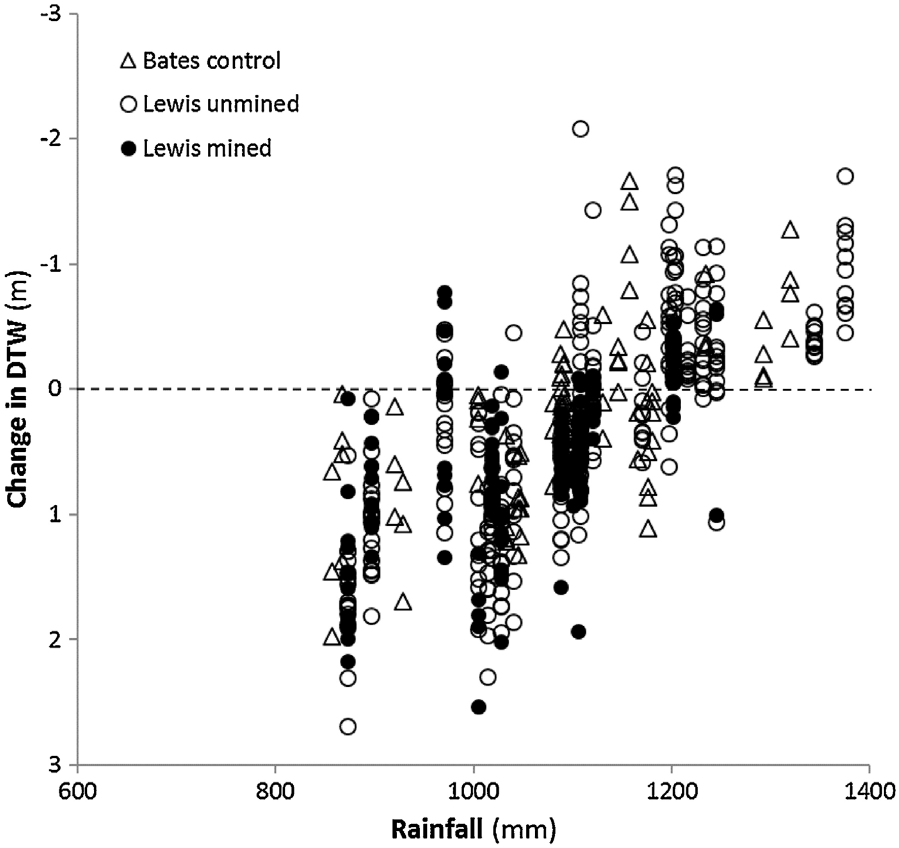

Fig. 5. Change in annual average depth to groundwater in relation to two-year averaged annual rainfall for piezometers in Bates and in mined or unmined parts of Lewis catchment. The dotted line divides occurrences of groundwater rise (negative values) from occurrences of groundwater fall (positive values).

flows, in the absence of any disturbance, cannot be determined. However, a more conservative estimate of the flow response to mining was calculated based on the slope and intercept at the lower end of the 95% confidence interval from the original regression (Eq. (2)):

$$Q_L = 0.66 * Q_B - 12.7 \tag{2}$$

For either set of estimates, there was minimal response in the first full year of mining in 1997, but the response increased steeply from 1998 (Table 2). Using Eq. (2), the estimated peak response to mining was approximately 225 mm or 18% of rainfall in the year 2000 (Table 2), with minimal further mining response from around 2008.

Annual streamflow response to mining in Lewis using the more conservative estimate from Eq. (2) (Table 2) was closely and linearly associated with the size of the groundwater discharge area (Fig. 8a). A similar linear relationship was also observed between the response to thinning treatment in Hansen, as estimated by Robinson et al. (1997), and the size of the groundwater discharge area each year estimated in the current study. Results from two further studies undertaken in jarrah forest catchments, one completely cleared for pasture development (Ruprecht and Schofield, 1989) and the second subject to a clearfell treatment (Bari et al., 1996), indicate a general pattern consistent across all of these land use types (Fig. 8a). Streamflow response to mining also showed a linear but inverse relationship with catchment average LAI (Fig. 8b). The pattern of response again could not be distinguished from the response to thinning in Hansen catchment.

4. Discussion

The response to mining in this study involved an initial increase in streamflow, peaking approximately four years after mining entry into the catchment, followed by a return to pre-mine flows after about 11 years. The peak response was estimated to be approximately 225 mm or 18% of rainfall, which was comparable to the peak response in Hansen of approximately 300 mm or 23% of rainfall reported by Robinson et al. (1997). The response in Lewis is toward the upper end of the range of responses reported for other mined catchments of 8–23% (Bari and Ruprecht, 2003; Croton and Reed, 2007; Ruprecht and Stoneman, 1993), and similar to responses to forest thinning (8–18% of rainfall: Bari and Ruprecht, 2003), but lower than the peak of 32% of rainfall in response to complete clearing (Ruprecht and Schofield, 1989). The duration of response of about 11 years in this study is also comparable to catchment logging and thinning studies in the jarrah forest, which show a return to pre-treatment flows after 12–15 years depending

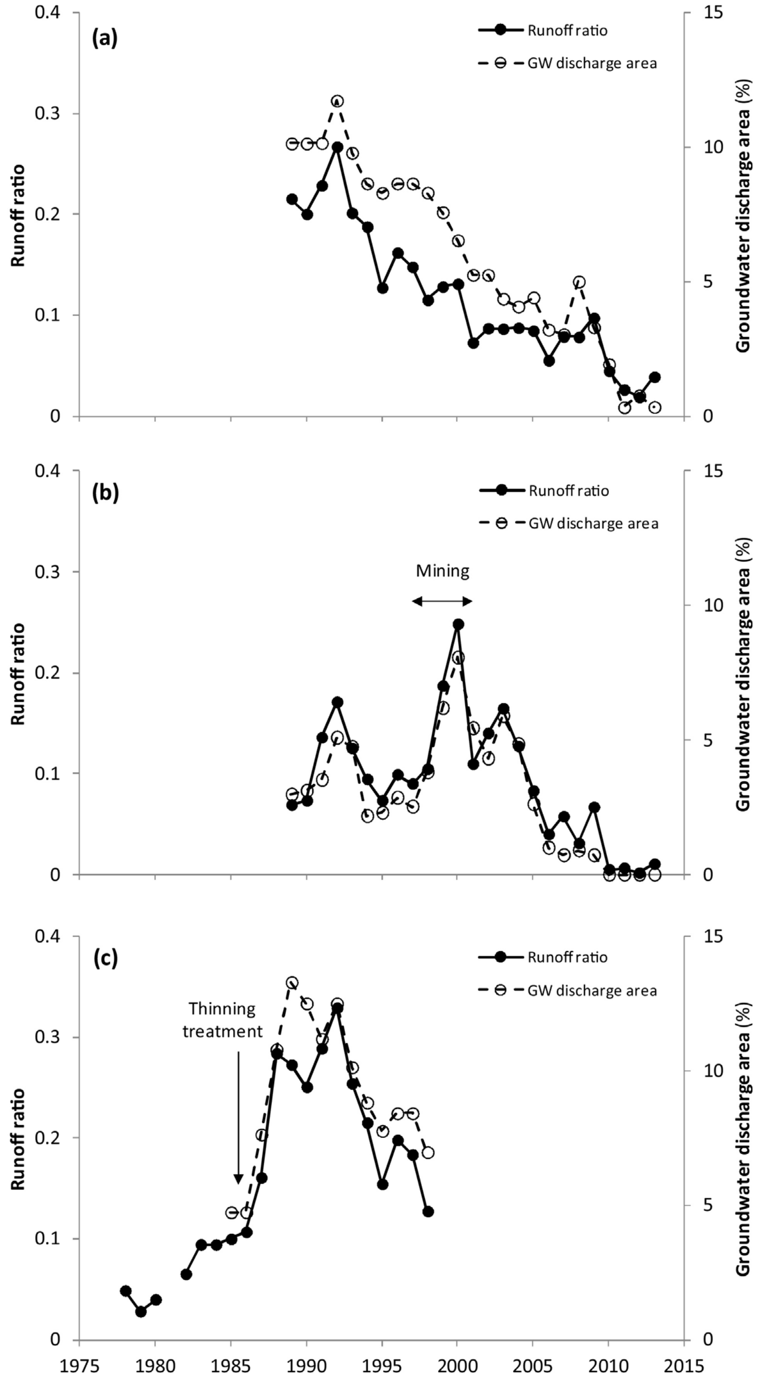

Fig. 6. Annual rainfall runoff coefficient, and estimated groundwater discharge area expressed as a percentage of the catchment area, for (a) Bates (control), (b) Lewis (mined) and (c) Hansen (thinned) catchments.

on the rate of vegetation regrowth (Bari and Ruprecht, 2003; Ruprecht and Stoneman, 1993).

Both the streamflow response to mining and the annual runoff ratio in Lewis were very closely related to the size of the groundwater discharge area (Figs. 6, 8). The same behaviour was evident for Hansen catchment in which forest LAI was reduced to levels comparable to Lewis (Fig. 2) but without the disruption to the upper regolith. In addition, the relationship between the annual runoff ratio and the groundwater discharge area from these two treated catchments was similar to the control catchment Bates (Fig. 6). More broadly, the responses to mining and intensive thinning reported in this study are indistinguishable from jarrah forest catchments subject to complete clearing, or clearfelling and regeneration (Fig. 8a). These results confirm the earlier conclusion of Ruprecht and Schofield (1989) that the 'permanent groundwater system is instrumental in controlling streamflow response' and are

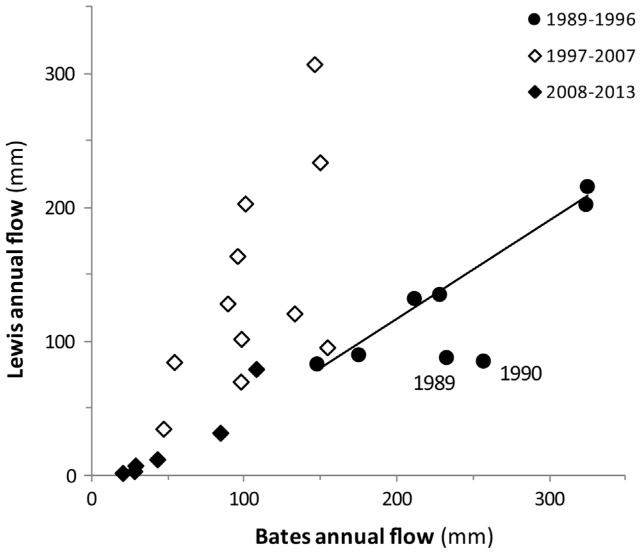

Fig. 7. Relationship of annual streamflow between Lewis (mined) and Bates (control) catchments in the pre-mining period (1989–1996), in the mining and early post-mining phase (1997–2007) and later post-mining phase (2008–2013). Details of the fitted regression (Eq. (1)) are given in Section 3.4; the two years of 1989 and 1990 not used in the regression are indicated.

consistent with more recent research that has reaffirmed the key role that groundwater storage plays in streamflow generation in jarrah forest catchments (Hughes et al., 2012; Kinal and Stoneman, 2012).

Groundwater responded to the combined but opposing influences of rainfall and forest LAI. At the local scale in this study, changes in the annual average depth to groundwater were positively related to rainfall. A threshold rainfall (on a two-year averaged annual basis) at which groundwater levels were maintained was found to be 1100–1200 mm. This is within the range of threshold rainfalls of 1050–1400 mm estimated for whole catchments by Hughes et al. (2012), who also postulated that differences in the threshold between catchments was likely related to forest management and forest density. Conversely, reductions in LAI led to increased recharge, as a result of decreases in transpiration and interception losses (Ruprecht and Schofield, 1989). Importantly, there was no significant effect of mining in the groundwater model developed here, indicating that neither disruption of the upper regolith nor the post-mining vegetation that was re-established altered the fundamental factors influencing the amount of recharge. Similarly,

Table 2
Streamflow response in Lewis catchment due to mining and rehabilitation, estimated using Equation (1) and Equation (2) (see Section 3.4).

Year	Rainfall (mm)	Measured flow (mm)	Eq. (1)		Eq. (2)	
			Predicted flow (mm)	Measured − predicted flow (% rainfall)	Predicted flow (mm)	Measured − predicted flow (% rainfall)
1991	1488	202	209	−0.5	201	0.0
1992	1261	215	210	0.5	202	1.1
1993	1078	135	138	−0.3	138	−0.3
1994	950	90	99	−0.9	103	−1.4
1995	1130	83	78	0.4	85	−0.2
1996	1333	132	126	0.5	127	0.3
1997	1061	95	83	1.1	89	0.6
1998	1154	121	67	4.6	75	4.0
1999	1252	234	80	12.3	86	11.8
2000	1238	307	77	18.6	84	18.0
2001	771	84	9.0	9.8	23	7.9
2002	1169	164	40	10.6	50	9.7
2003	1234	203	43	12.9	54	12.1
2004	1010	128	35	9.2	46	8.1
2005	1229	102	42	4.9	52	4.0
2006	871	35	3.8	3.5	18	1.9
2007	1214	70	41	2.3	52	1.5
2008	1027	32	31	0.02	43	−1.1
2009	1188	79	49	2.6	59	1.7
2010	605	2.9	−10	2.2	6	−0.5
2011	1140	7.1	−10	1.5	6	0.1
2012	1076	1.5	−16	1.6	1	0.1
2013	1130	11.7	1	1.0	16	−0.4

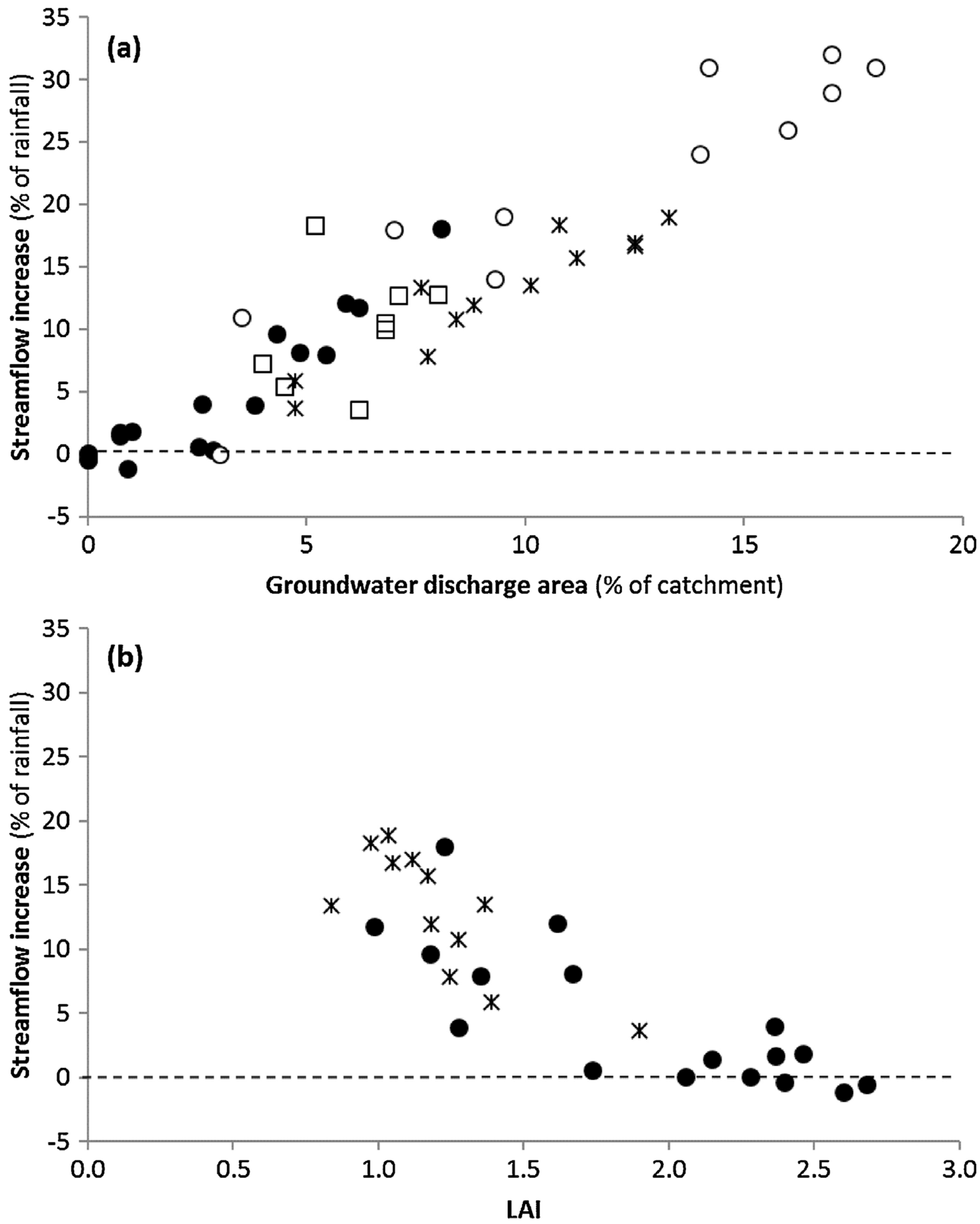

Fig. 8. Estimated streamflow response to mining in Lewis using Eq. (2) (●) and thinning in Hansen (*) (Robinson et al., 1997) in relation to (a) the size of the groundwater discharge area and (b) catchment average LAI. Also shown in (a) are streamflow responses reported for two other catchment studies in the jarrah forest: March Rd (□) (Bari et al., 1996) and Wights (○) (Ruprecht and Schofield, 1989).

Hughes (2012) who investigated the time delay between individual rain events and groundwater recharge for piezometers in mined and unmined jarrah forest catchments, found that only depth-to-water was a significant predictor.

Bauxite extraction clearly disturbs the upper regolith and eliminates any indurated layer present. If there was a permanent change to streamflow generation, then it should have been detectable in Lewis where more than half of the catchment (encompassing almost all upland areas) was mined and rehabilitated (Fig. 1). To the contrary, this study found no discernible difference in the streamflow response in Lewis compared to other land use types which cause little disturbance to the regolith. These findings contradict the notion that shallow sub-surface flow or throughflow is the dominant mechanism for the transfer of rainfall to streams in these catchments, whereby infiltrating rainfall 'perches on the clay B horizon and flows downslope to discharge to streams' (Bari and Ruprecht, 2003). Such a shallow sub-surface pathway would be expected to be disrupted by the mining process and be revealed as a departure in streamflow response when compared with other land use types, but this was not the case. Observations of ephemeral saturation above a duricrust or in fine-textured soils are put forward in support of this view (Ruprecht and Stoneman, 1993), but perching may not be as extensive across the landscape as implied. Robinson et al. (1997), for example, report true perching in only two of a total of 23 shallow piezometers across four catchments, and other studies highlight significant vertical fluxes to depth explained by the presence of preferred flow channels (McFarlane and Williamson, 2002; Turner et al., 1987a). The magnitude of lateral movement of infiltrated rainfall, or interflow, will be determined by the hydraulic conductivities of the upper and impeding layers, topographic gradient and the thickness of the saturated lens (Jackson et al., 2014; McFarlane and Williamson, 2002). Application of the formula provided by Jackson et al. (2014) using typical values for jarrah forest surface soils and subsoils (Sharma et al., 1987) suggests maximum interflow distances to be in the order of tens of metres. Jackson et al. (2014) extend their analysis to conclude that a catchment may be divided into zones, with significant interflow likely to be limited to the valley floor and immediate surrounds. None of these areas were directly impacted by mining in Lewis (Fig. 1), as is typically the case for bauxite mining across the jarrah forest (Koch, 2007a).

The notion of downslope interflow as a dominant process may have arisen from earlier research that showed that contributions to streamflow in these catchments are dominated by shallow throughflow, and that deep groundwater contributions are relatively small (Stokes and Loh, 1982; Turner et al., 1987b). A drying climate in the south west of Australia, however, has challenged this notion by placing greater emphasis on the role of deep groundwater (Hughes et al., 2012; Kinal and Stoneman, 2012; Hughes and Vaze, 2015). The present study expands upon our understanding by indicating that the facilitated shallow throughflow component is likely to be largely confined to the valley floor and immediately adjacent lower slopes. The role of the remainder of the catchment in a hydrological sense is in controlling recharge and hence overall catchment storage. Additional recharge within and downslope of cleared mine pits is clearly visible (Figs. 3 and 4) and groundwater beneath rehabilitated pits responds to rainfall and LAI in the same way as unmined forest areas, from which it is concluded that mining does not fundamentally alter the processes leading to streamflow generation in this environment.

Ongoing declines in groundwater storage in both Lewis and Bates catchments can be expected if the lower-than-average rainfall conditions experienced in the post-mining period persist. Since groundwater storage is negatively associated with forest LAI, declines are likely to be more rapid under mined and rehabilitated areas where the post-mining LAI has plateaued at a higher level than the pre-mine forest (Fig. 2). There are early indications that this may have already occurred in mid-slope and valley locations (Fig. 4). In the case of Lewis catchment, this is unlikely to significantly influence streamflows in the short term as groundwater disconnection is well advanced and streamflows are already small. However, a slower return to a groundwater connected state and associated higher flows relative to the unmined alternative may be anticipated should a wetter rainfall regime return in the future. The relatively higher total overstorey and understorey LAI in rehabilitated parts of this catchment are comparable to other published estimates in bauxite mine rehabilitation of a similar establishment era (Macfarlane et al., 2010) and are due to both higher tree densities than unmined forest and a substantial understorey component. This reflects tree and understorey seeding protocols of the time which were notably higher than current seeding rates. Tree and leguminous understorey seeding rates were substantially decreased from 2000 (Alcoa, unpublished data), and this is evident in the slower recovery in LAI in 2001 rehabilitated areas when compared with 1998/1999 rehabilitation areas (Fig. 2b). For existing stands such as in Lewis, silvicultural treatment such as thinning and fuel-reduction burning may be considered in managing the longer-term development of rehabilitated areas (Grigg and Grant, 2009) and associated hydrological effects.

Declines in groundwater levels and runoff coefficients occurred across both mined and control catchments over the course of the study (Fig. 3) and in Lewis, groundwater at the catchment outlet showed increasing 'disconnection' from the valley floor after the record drought year in 2010 (Fig. 6). Kinal and Stoneman (2012) also reported the abrupt and substantial drop in runoff coefficient associated with such a change in catchment hydrological state. This provides a possible explanation for the anomalous years of 1989 and 1990 in the comparison of streamflows in Lewis and Bates in the pre-mine period (Fig. 7). Streamflow characteristics in Lewis in the two years prior suggest that connectivity was weak, consistent with relatively lower rainfall and high catchment LAI at the time. In contrast, the groundwater discharge area in Bates is likely to have been comparatively larger during the same period, giving rise to contrasting hydrological states in the two catchments. Robinson et al. (1997) encountered a similar issue when comparing Lewis with Hansen in their study, describing Lewis as an unstable control and presenting estimates of treatment response by both a paired catchment approach (which is reported here) and by changes to the rainfall-runoff relationship in Hansen alone. This highlights the difficulties and potential problems of the paired catchment approach in this environment. For the present study, the relationship between Lewis and Bates was affected only when flows in both catchments were small and the error in prediction correspondingly small. However, any future investigation into the longer-term effects of mining in Lewis catchment will be unable to use the same paired catchment approach. More generally, drying conditions and further decreases in streamflow are likely in the south west of Western Australia (Silberstein et al., 2012) and future catchment studies in the jarrah forest will need to closely consider the efficacy of the paired catchment approach in the light of progressive groundwater disconnection.

5. Conclusions

Mining for bauxite in the jarrah forest caused a peak response in catchment streamflow of 225 mm or 18% of rainfall before returning to pre-mine levels 11 years after mining commenced. Changes in streamflow were closely associated with an expansion and subsequent contraction of the groundwater discharge area in the valley floor, which in turn was primarily driven by changes in LAI and rainfall. The response to mining could not be distinguished from responses to other catchment disturbances which do not disrupt the regolith including forest thinning and clearing, indicating that shallow subsurface flow processes, considered to dominate streamflow generation in jarrah forest catchments, do not extend beyond the valley floor and immediately adjacent slopes. The effects of climate and especially very dry years were evident in streamflow declines in both mined and control catchments during the period of records. In Lewis catchment where LAI of rehabilitated areas has risen above pre-mine levels, silvicultural treatment such as thinning and fuel-reduction burning may be considered in managing vegetation development and associated longer-term hydrological effects of mining.

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