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Smoke on the water: Can riverine fish populations recover following a catastrophic fire-related sediment slug?

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Abstract In early 2003 a series of large, wildfire-related sediment slugs occurred in streams in the south-eastern Australian alpine region. Back-pack and boat-mounted electrofishing were used to measure changes in riverine fish fauna after one particularly large sediment slug which started in an upland stream and then travelled downstream through 200 km of third and fourth order stream. Twelve impact sites and eight control sites were surveyed where there were previous data on fish populations. The sites were surveyed directly after the sediment slug had passed and then 12, 24 and 36 months after. Immediately after the sediment slug, fish abundances fell by between 95–100% at four impact sites in the upper reaches of the study area, primarily due to the effect of low-dissolved oxygen levels. Twelve months later fish numbers were still decreased in the upper catchment but showed signs of recovery after 24 months. Further downstream, where water quality was not as severely affected by the sediment slug, the effects on native fishes were less apparent. The circumstances of these events represented a unique opportunity to obtain baseline data regarding the effects of post-fire disturbances on fish, and their time to recovery. If fire were to occur in catchments where endangered species exist, our results suggest that actions such as translocation may be required to ensure the long-term survival of threatened species.

Key words: Australia, blackfish, dissolved oxygen, fire, Ovens River, sediment.

INTRODUCTION

South-eastern Australia is generally regarded as one of the world's most fire-prone environments because of its high temperatures, low rainfall and flammable native *Eucalypt* forests (Wareing & Flynn 2003). Many *Eucalypts* have evolved to cope with wildfire, and indeed many species need fire to complete their life cycles (Pyne 1991). Moreover, there is evidence to suggest that the arrival of the Aboriginal people 40 000 to 60 000 years ago dramatically increased the frequency of fires. Aborigines utilized fire to create patches of fresh ground cover that attracted animals such as kangaroos and wallabies, which could then be more easily hunted (Pyne 1991). As a result, fire has long been a natural part of the Australian landscape.

The effects of wildfires on plants and terrestrial animals are instantly visually apparent. What is not so well recognized is that wildfires can also profoundly influence aquatic ecosystems, both directly and indirectly. Direct effects include increased water temperatures (Minshall & Brock 1991; Hitt 2003), increases in stream pH (Cushing & Olson 1963) and increases in nutrients (in particular nitrogen and phosphorus) from smoke and ash inputs (Beschta 1990; Bayley *et al.* 1992; Earl & Blinn 2003). Occasionally, freshwater fish have died as a direct result of fire, but these cases have usually been in small isolated waterways, often affecting areas less than 1 km in length, and usually involving only small numbers of fish and partial elimination of a population (Cushing & Olson 1963; McMahon & de Calesta 1990). In any case, direct effects of fire are likely to be confined to smaller first and second order streams where the smaller volume of water makes changes in water temperature and chemistry more likely to occur.

The indirect impacts of fire on aquatic systems arise from the effects on the surrounding landscape. These include increased water temperatures caused by a lack of riparian vegetation (Albin 1979; Minshall *et al.* 1997; Royer & Minshall 1997), silting of pool habitats (Benda *et al.* 2003), increased inputs of nutrients from burnt material (Minshall *et al.* 1989; Bayley *et al.* 1992; Minshall *et al.* 1997) and altered flow regimes caused by the increased uptake of surface and ground water by young plants. However, possibly the greatest threat to aquatic fauna comes from post-fire rainfall and the increased sediment loads that accompany increased run-off from recently burnt ground (Beschta 1990; Novak & White 1990; Beaty 1994; Rieman *et al.* 1997; Benda *et al.* 2003; Meyer & Pierce 2003).

During a wildfire, a variety of processes increase the probability of erosion, the most obvious being the loss of vegetative ground cover and leaf litter which help stabilize soils. However, soil structure can also be

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damaged by the combustion of organic matter within the soil and the consequent reduction in soil binding. In addition, intense heating can cause some soils to become hydrophobic, therefore creating a higher probability of surface runoff (Wondzell & King 2003). As such, loads of suspended sediment in streams are often at their highest after flash flooding arising from rainfall immediately after wildfire (Novak & White 1990; Benda *et al.* 2003; Miller *et al.* 2003). Such heavy rainfall often causes debris flows (sediment slugs) that contain tremendous amounts of sediment, and it is these flows that are generally attributed with decimation of fish populations.

The severity of such sediment slug events depends primarily on the gradient of the land, the intensity of the fire and the intensity of the rainfall event. In a forested area of Montana USA, such a debris flow almost eliminated populations of brown (*Salmo trutta*) and rainbow (*Onchorhynchus mykiss*) trout from the affected area of stream (Novak & White 1990). Similarly, populations of brook trout *Salvelinus fontinalis* and rainbow trout were completely extirpated from streams in the south-western USA following fire-related sediment slug events (Rinne 1996). Although a relatively substantial body of published data can be found on such events for streams in North America, usually involving salmonids at relatively high altitudes, there is no comparative published information regarding such events from Australian streams and the subsequent effects on Australian native fish species.

In early 2003 bushfires burnt 1.73 million hectares of forest and farmland in south-eastern Australia. Post-fire rainfall then washed large amounts of sediment and ash into the headwaters of some of the region's most pristine streams. This study was opportunistic and undertaken to assess the damage to fish populations in the Ovens and Buckland rivers caused by a large, fire-related sediment slug.

METHODS

Study area

The reach of river impacted by the sediment slug is in northern Victoria, in the south-east of Australia (Fig. 1). The sediment slug event began in the Dingo Creek, a second order tributary of the Buckland River. On 26 February 2003, approximately 150 mm of rain fell in 1 h over an isolated area of less than 100 ha in the upper Dingo Creek catchment. Large areas of the upper catchment were subjected to severe burning by wildfire several days before (Fig. 1). The rain caused an immense surge of debris and sediment, which travelled downstream the length of the Buckland (3rd order) and much of the Ovens (4th order) rivers and subsequently reduced water quality and deposited large amounts of sediment as it subsided. Twelve impact sites and eight control sites were sampled in the Buckland, Ovens and adjacent King river catchments (Fig. 1). The sites were classified into: upland impact – which were generally shallow (<1 m) and steep gradient reaches of stream (sites i1–i6); lowland impact – which were relatively deep (>1 m (and up to 5 m)) and low gradient reaches of stream (sites i7–i12); and control sites which had similar geomorphologic characteristics, but were located either in an adjacent catchment or in an area not affected by the sediment slug (c1–c8).

The Ovens catchment is inhabited by three large-bodied federally endangered percichthyids: Murray cod *Maccullochella peelii*, trout cod *Maccullochella macquariensis* and Macquarie perch *Macquaria australasica*. In addition, the native species golden perch *Macquaria ambigua*, two-spined blackfish *Gadopsis bispinosus*, river blackfish *Gadopsis marmoratus,* Australian smelt *Retropinna semoni*, flat headed galaxias *Galaxias rostratus*, southern pigmy perch *Nannoperca australis* and mountain galaxias *Galaxias olidus* are present (Koehn 2006).

Recently obtained data on fish diversity and abundance for the Ovens catchment were gathered between 1997 and 2002, allowed a quantitative assessment of the impact of the post-fire sediment slug on fish populations immediately after the fire and over the following 3 years.

Water quality

An automatic flow-measuring device located approximately 30 km downstream from the source of the sediment slug (Harris Lane logger on the Buckland River) measured river discharge as the flood pulse passed. Dissolved oxygen was also measured by a data logger installed by the authors in the Ovens River at Tarrawingee, about 70 km downstream from the source of the sediment slug. Dissolved oxygen and turbidity were also measured using spot measurements taken at different sites within the study area during sampling. In the period immediately after the sediment slug and at 12, 24 and 36 months following, deposited sediment was also measured in the affected zone by undertaking a series of transects. Ten sediment transects (located 10 m apart) were taken at both sites i3 and i4, with sediment depth measured at 1 m intervals across the river.

Data collection

Pre-sediment slug fish abundance data were compiled from sites that were sampled between 1997 and 2002

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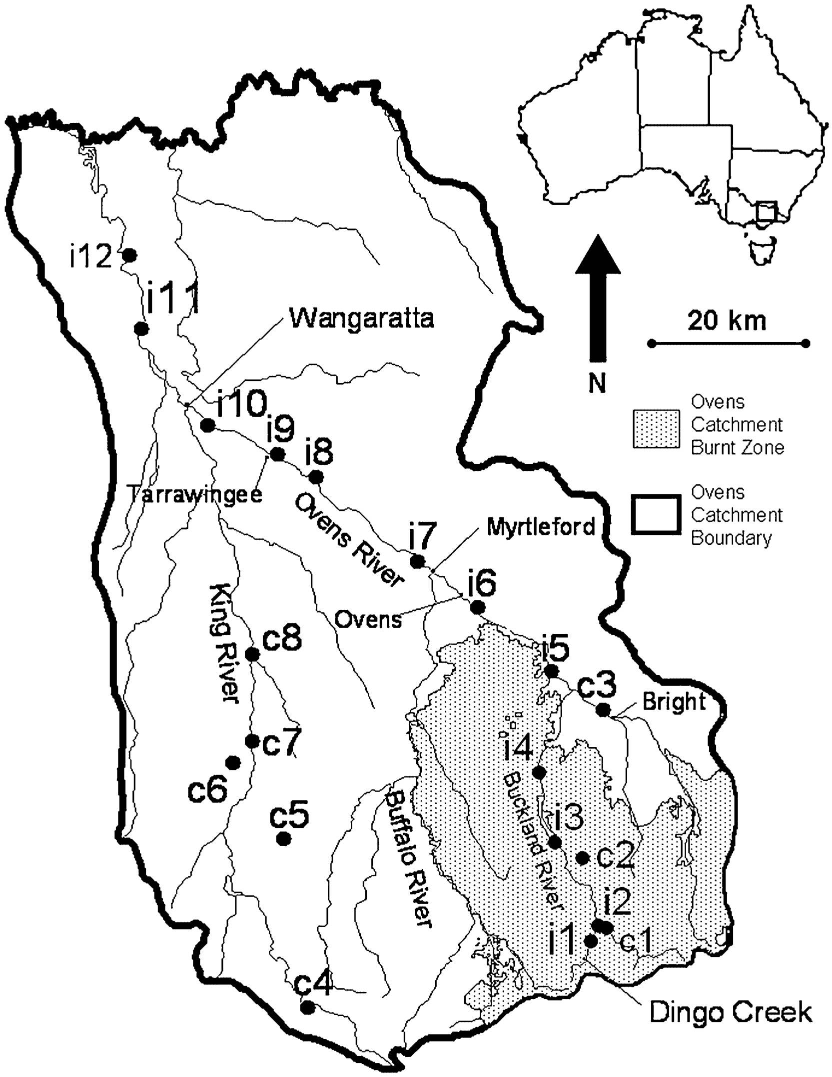

Fig. 1. Ovens and Buckland rivers catchments in south-eastern Australia indicating sites surveyed. Note: 'i' indicates impact, and 'c' indicates control. Sites C2, C5 and C6 are located on tributaries of the major streams shown.

as components of different study programs undertaken by the Arthur Rylah Institute (Table 1). In order to obtain information on the immediate impact of the sediment in early 2003, five sites in the upland section of the study area were sampled once by repeating the electrofishing methodology from previous surveys at these sites. These sites could be sampled as the turbid water had flushed downstream. However, the high turbidity from the sediment slug made it impractical to resample lowland sites due to an inability to see fish in the water. Following the partial sampling undertaken in early 2003, all sites were sampled in January 2004, January 2005 and again in January 2006.

Upland impact sites were all sampled using backpack and bank mounted electrofishing, while lowland impact sites were all sampled using boat-mounted electrofishing. Fishing effort was standardized between years at each site (by area surveyed) to ensure a relevant annual comparison. Sites were determined according to the previous area sampled. Although this differed between sites, the techniques recorded fish captured per unit of time over a defined site area by maintaining site-specific methodology between sampling years. A BACI (Before, After, Control, Impact) (Underwood 1992) type experimental design was chosen to compare the effects of the sediment slug using recent fish survey data from a number of sites along the Buckland and Ovens rivers. A total of 12 impact and eight control sites were sampled to measure post-slug impacts and recolonization rates of fish assemblages in the Ovens, Buckland and King rivers and their tributaries (Fig. 1). Due to their inconsistent distribution and often high sampling densities, the small (<100 mm) schooling Australian smelt were

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Table 1. Survey programs from which pre-fire fish abundance data was sourced

Pre-fire data	Survey program	Method
i1	No Previous data	N/A
i2	RFA	Single pass electrofishing
i3	RFA	Single pass electrofishing
i4	RFA	Single pass electrofishing
i5	Blackfish assessment	Double pass electrofishing
I6	SRA	Single pass electrofishing at timed intervals
i7	SRA	Single pass electrofishing at timed intervals
i8	SRA	Single pass electrofishing at timed intervals
i9	SRA	Single pass electrofishing at timed intervals
i10	SRA	Single pass electrofishing at timed intervals
i11	SRA	Single pass electrofishing at timed intervals
i12	SRA	Single pass electrofishing at timed intervals
c1	No previous data	N/A
c2	RFA	Single pass electrofishing
c3	Blackfish assessment	Double pass bank-mounted electrofishing
c4	SRA	Single pass electrofishing at timed intervals
c5	RFA	Single pass electrofishing at timed intervals
c6	RFA	Single pass electrofishing
c7	SRA	Single pass electrofishing
c8	SRA	Single pass electrofishing
Pre-fire Data	Survey Program	Method
All sites	Repeated as above	As above

For further information regarding pre-fire data obtained from surveys as followed: Blackfish Assessment – Koster (2004); RFA (Regional Forest Agreement) – Raadik *et al.* (2000); SRA (Sustainable Rivers Audit) – MDBC 2004).

noted as present/absent only, and as such have been excluded from the analysis.

Statistical analysis

As the data consisted of counts, a Poisson regression modelling approach using a log link was selected as the probability model for the data. As repeated measures were taken from each of the 20 sites over time, a generalized linear mixed-effect modelling (GLMM) approach was used with individual site fitted as the random effect (effect on the variance of fish counts). Fixed effects (effects on mean fish counts) in the model were river section (upland impact, lowland impact and control) and time (pre-slug, post-slug 1 m, post-slug 12 m, post-slug 24 m, post-slug 36 m). The response variable of interest was total fish abundance and five models were fitted of increasing complexity to explain the variation in this response. These were:

- Constant model – counts did not vary by section or time
- Section model – counts varied between river sections but not by time
- Time model – counts varied by time but not by river section
- Section + Time – counts varied by time with an additive effect of river section
- Section × Time – Counts varied by time and river section independently

The relative support for each of these models was assessed by calculating Akaike's information criterion (AIC), corrected for small sample size (AIC_c) (Anderson *et al.* 1994). AICc values were rescaled as differences between the model and the model with the lowest AICc value. The likelihoods of the models, given the data were calculated as

$$P(M_k|y) = \frac{e^{(-0.5\times\Delta\mathrm{AICc}_k)}}{\sum_{k=1}^{5} e^{(-0.5\times\Delta\mathrm{AICc}_k)}} \quad (1)$$

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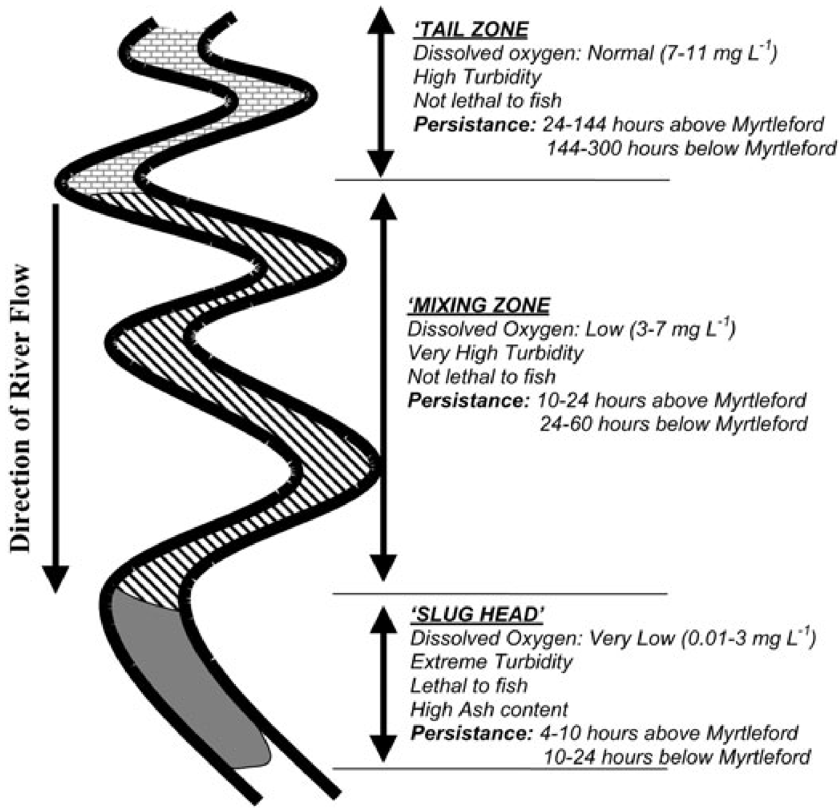

Fig. 2. Proposed model of the composition of sediment slug. Shaded areas represent the 'slug head', diagonal lines represent the 'mixing zone' and bricked areas represent the 'tail zone'. Note that the persistence (the amount of time for any particular slug section to pass a set point) above and below the township of Myrtleford is different. This is primarily due to a shift in the stream gradient from upland stream to lowland stream in this vicinity. Note also that there was mixing between the three slug sections.

Where $P(M_k|y)$ is the likelihood of model M_k given data y (Hoeting *et al.* 1999).

Initial modelling revealed the presence of overdispersion in the count data. Hence quasi-likelihood methods were used to fit the GLMM models. The likelihood for these models was approximated using a Laplacian approximation to the full maximum likelihood. Hence a quasi-likelihood approximation to AIC_c, adjusting for overdispersion, was used for model selection ($QAIC_c$) (Anderson *et al.* 1994). Parameter estimates were obtained by model averaging to account for uncertainty in model selection using the estimates of model likelihood (eqn 1) as the model weights.

RESULTS

Water quality

Water quality in the Buckland and Ovens rivers was severely degraded as a result of the post-fire rain event. Sediment and ash were washed into watercourses, quickly forming into a sediment slug. For the purpose of this study, we have proposed a model that described the structure of the slug based on water quality parameters that we observed. The model divided the event into three sections of varying intensity (Fig. 2), and may also be useful for describing other point source pollution events in the riverine environment. The 'head' of the sediment slug (Fig. 2) was the most severely degraded section where water was extremely turbid and there was low-dissolved oxygen. This section of the sediment slug was lethal to fish. Water quality began to recover in the 'mixing' section of the sediment slug, and while turbidity and dissolved oxygen had not returned to pre-slug levels this section of the sediment slug did not appear to be lethal to fish. The final section of the sediment slug was the 'tail' zone where most water quality parameters had returned to pre-slug levels except for turbidity which remained relatively high. The 'tail' zone of the sediment slug was not directly lethal to fish; however, sediment deposited by the tail of the sediment slug could have indirect adverse effects.

Water quality parameters showed large fluctuations as a result of the sediment slug. Thirty kilometres downstream of the source of the sediment river dis-

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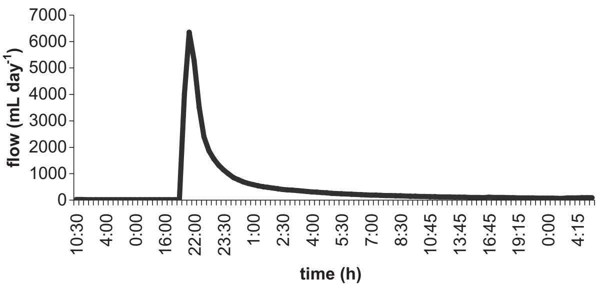

Fig. 3. Flood pulse measured at an automatic flow measuring device 30 km downstream from Dingo Creek on 26 February 2003. Note the flow rate before the flood peak of 6400 mL day^{-1} was between 4 and 5 mL day^{-1} (Source: Victorian Data Resources Data Warehouse).

charge peaked at approximately 6400 mL day^{-1} as the flood pulse passed (Fig. 3). Suspended sediment loads of up to 57 000 mg L^{-1} (or 5.7% solids) were recorded in the Buckland River at the Porepunkah water supply off-take (approximately 300 m upstream from site i4) at the head of the slug (Leak *et al.* 2003). From these data, the amount of sediment moving down the river at this site was approximately 4 t s^{-1}, or 2400 t passing this point in 10 min at a flow of 6 000 mL day^{-1}. In contrast, the flow rate before the storm was between 4 and 5 mL day^{-1} (Fig. 3) with an associated sediment load of approximately 1.15 kg s^{-1} (0.00115 t s^{-1}).

Point dissolved oxygen measurements were recorded at several sites (i6, i7, i8) above Tarrawingee (a township located approximately 70 km downstream from the source of the sediment) and were extremely low (<1 mg L^{-1}) at the head of the sediment slug. A data logger (fitted with a YSI 5739 oxygen probe) installed in the river at Tarrawingee measured dissolved oxygen as low as 0.2 mg L^{-1} where it remained for approximately 12 h, after which it began to rise, peaking at 4 mg L^{-1} (Fig. 4). Sediment deposited on the probe prevented readings returning above 4 mg L^{-1}. Dissolved oxygen in the Buckland and Ovens rivers generally ranged 7–10 mg L^{-1} (J. O'Connor, unpubl. data 1999). The head of the sediment slug reached Tarrawingee about 6 days after the rain event. Downstream of Tarrawingee the lower gradient of the Ovens River began to slow the velocity of the sediment slug. The sediment slug took a further 5 days to travel the 30 km from Tarrawingee to Wangaratta by which time turbidity had decreased dramatically. Water electrical (specific) conductivity (EC) increased substantially at the height of the slug from 40 to 180 μS cm^{-1} before returning to normal at the tail of the slug. However, this increase in EC, while large, remained well within the tolerance limits of all fish species.

Large amounts of sediment were deposited in the Buckland and Upper Ovens rivers. In the area from the source to approximately 40 km downstream all pool and run areas were subject to a thick layer of fine sediment. Indeed, some pools were almost filled with this sediment to a depth of 1.5 m. Sediment transects indicated that average sediment depth at site i3 (following the sediment slug) was 8.37 cm in 2003 (SE 0.97 cm) and 2.90 cm in 2004 (SE 0.78 cm). Average sediment depth at site i4 was 8.99 cm in 2003 (SE 1.77 cm) and 2.07 cm in 2004 (SE 0.908 cm). Apart from some thick (i.e. 50 cm) layers of sediment high on the banks (>1 m out of the water), sedimentation was at pre-slug levels by 2005. After the initial sediment inputs we measured no additional sediment input into the study streams.

During the sediment slug event, fish were found dead and struggling at the surface as far downstream as Tarrawingee (authors' observations 2003). Although we only observed relatively small numbers of fish which had actually died (Table 2), it is likely that other dead fish were present but buried in the thick sediment. Large numbers of freshwater crayfish and yabbies were also observed walking out of the water, and in some areas hundreds of individuals lined the banks. No dead fish were reported from control sites.

Fish sampling – general

A total of 5701 fish were captured on the sampling occasions that encompassed the pre-fire surveys (1997–2003), and the post-fire surveys (February 2003–January 2006). These comprised 4834 native fish and 867 introduced fish. The most abundant native fish species was the mountain galaxias *G. olidus* which comprised 42% of the total catch while the least abundant native fish was the flat-headed gudgeon, of which only three individuals were collected. Seventy-three federally endangered trout cod were also caught. The most abundant introduced fish was common carp with 5.7% of the total catch. Salmonids (brown trout *S. trutta* and rainbow trout *Oncorhynchus mykiss*) comprised 7% of the total catch (Table 3).

Statistical analysis

Model selection indicated that total fish abundance varying both by section and time independently was the most likely model (Table 4). Hence, there was significant variation in the counts among river sections over time. The predicted mean counts in each section over time are given in Figure 5.

Upland impact sites (i1–i6)

The mean count for the upper impact sites decreased substantially immediately following the post-fire sedi-

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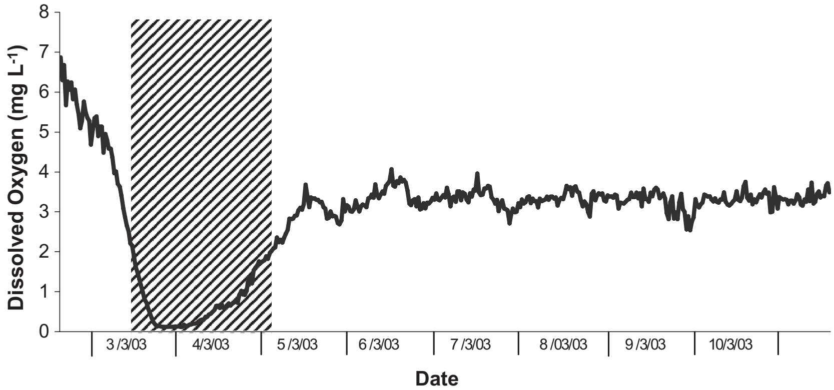

Fig. 4. Dissolved oxygen concentration in the Ovens River at Tarrawingee (70 km downstream from the source) before, during and after the sediment 'slug'. The shaded area indicates the 'head' of the slug passing downstream through the Tarrawingee area.

Table 2. Abundances of dead fish located during the sediment slug event

Species	Abundance
Gadopsis bispinosus	30
Maccullochella peelii peelii	4
Macquaria ambigua	1
Retropinna semoni	30
Hypseleotris spp.	2
Maccullochella macquariensis	1
Cyprinus carpio	>100

Table 3. Percentage abundances for fish captured during the current study (including pre-fire data)

	% Total catch
Native species	
Galaxias olidus	42.01
Gadopsis bispinosus	32.22
Gadopsis marmoratus	6.67
Maccullochella peelii peelii	2.14
Maccullochella macquariensis	1.28
Hypseleotris spp.	0.21
Macquaria ambigua	0.11
Nannoperca australis	0.11
Philypnodon grandiceps	0.05
Total	84.79
Exotic species	
Salmonids	7.00
Cyprinus carpio	5.68
Perca fluviatilis	2.37
Gambusia holbrooki	0.16
Total	15.21

ment slug, indicating there was a substantial impact upon these sites (Fig. 5). However, a gradual improvement in the mean count was observed over the following 3 years until in early 2006 mean counts were similar to the pre-sediment slug fish abundances.

Fish abundances decreased dramatically in the upland impact sites following the sediment slug (Fig. 6). Immediate post-fire sediment slug data indicated that a 100% reduction in fish abundances was recorded at sites i4 (decrease in fish abundance from 39 to 0) and i5 (decrease in fish abundance from 229 to 0). At site i3 a 90% reduction in fish abundance was recorded (decrease in fish abundance from 28 to 3). It is assumed that the three adult two-spined blackfish collected at this site immigrated into the site from the unimpacted East Buckland River (which is located approximately 50 m upstream) after the sediment slug had passed through. At site i6 only two juvenile Eastern gambusia *Gambusia holbrooki* were captured (decrease in fish abundance from 69 to 2).

In 2004, 12 months after the sediment slug, fish abundance at sites i3–i6 was beginning to increase, but remained lower than pre-fire (Fig. 7). However, 16 young-of-year (<75 mm) *G. bispinosus* were sampled from site i3. There were no large numbers of young-of-year *G. bispinosus* sampled at any other high-impact site in 2004. Sampling in early 2005 (24 months post sediment slug) indicated that fish abundance had increased in all upland impact sites, with abundances at some upland sites having returned to pre-bushfire numbers (Figs 6,7). In 2006 (36 months post sediment slug) the abundance of fish at all high-impact sites had returned to, or exceeded pre-sediment slug abundances.

Lowland impact sites (i7–i12)

The mean count in the lower impact sites did not show the same trend as the upper impact sites and, indeed, remained fairly constant throughout the study period, indicating there was little evidence of impacts from the sediment slug at these sites (Fig. 5).

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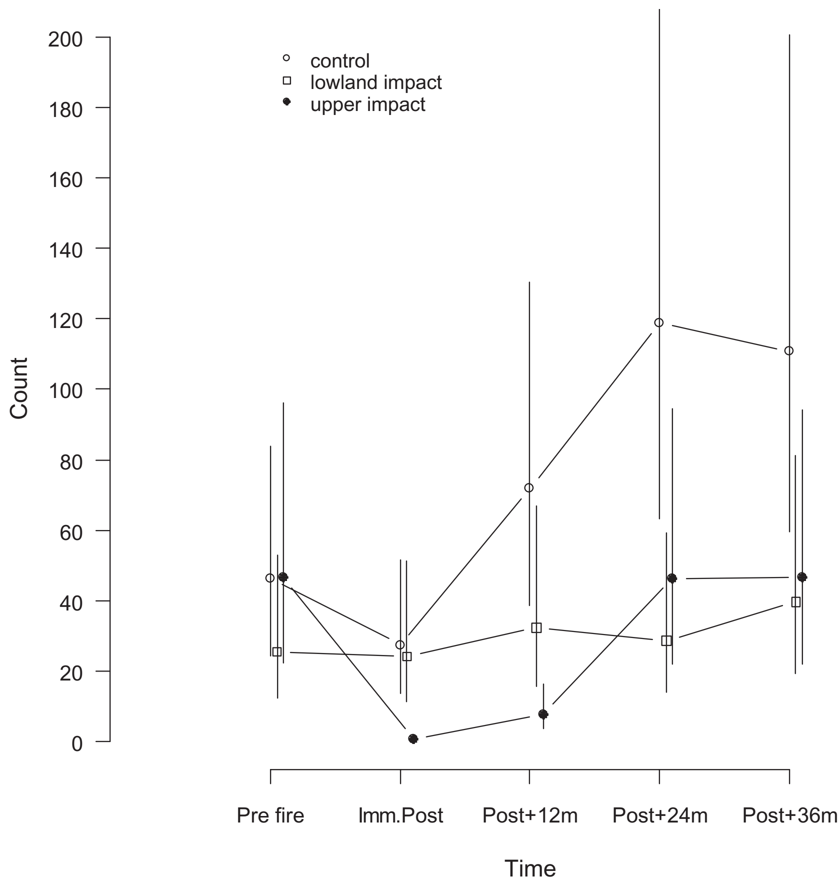

Fig. 5. Predicted total species counts in each river section pre- and post-sediment slug. Estimates and 95% intervals were obtained by model averaging using the five models in Table 4. Imm.Post, immediately post sediment slug; Post+12 m, 12 months post sediment slug; Post+24 m, 24 months post sediment slug; Post+36 m, 36 months post sediment slug.

Table 4. Results of the model selection procedure for five models fitted to the total species counts comparing section and time

Model	$QAIC_c$	Npar	$\Delta QAIC_c$	Likelihood
Section × time	441.2	16	0.0	1.00
Time	461.9	6	20.8	0.00
Section + time	465.5	8	24.3	0.00
Constant	532.1	2	90.9	0.00
Section	534.9	4	93.7	0.00

Site i7 was identified as the most upstream site where fish abundances were not reduced following the post-fire sediment slug. In the immediate post-fire sampling event 18 individual native fish and six exotic fish were recorded from this site, a slight increase in native fish abundance from previous surveys. No noteworthy changes in fish abundance were observed at any of the sites downstream of i7 (Fig. 6). In 2004 (12 months post sediment slug), 2005 (24 months post sediment slug) and 2006 (36 months post sediment slug), we did not determine any measurable impacts on fish abundance as a result of the sediment slug at the lowland sites (Fig. 7).

Control sites (c1–c8)

The mean count of the control sites showed large variations between years including a large decrease following the post-fire sediment slug; however, this decrease was not as dramatic as the decrease in mean fish count which was observed in the upper impact sites (Fig. 5).

There was high variability in fish abundance between years and locations at those control sites located within the King River catchment. Three of the control sites in the King River catchment showed large increases in fish abundances relative to pre-fire

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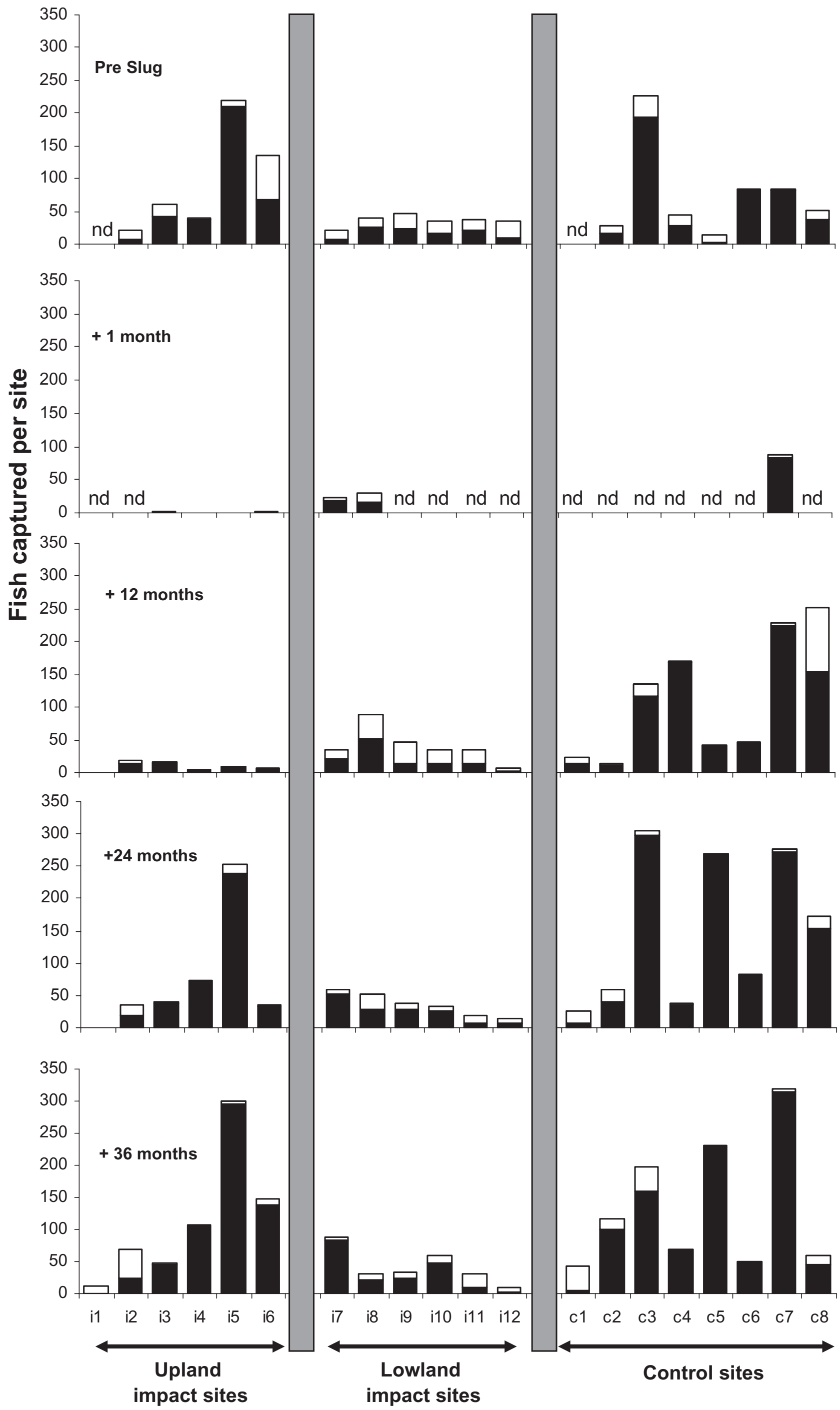

Fig. 6. Distribution of fish caught at each site over the sampling period. Dark portions represent native fish, and white portions exotic fish species. All sites were sampled by electrofishing. ND designates no data available (i.e. sample was not taken). Vertical bars separate sampling zones. Note that *y*-axis indicates total catch.

surveys, while one site showed a large decrease. However, no control site showed any decrease in fish abundance that was comparable to the 90–100% decreases associated with the high-impact sediment slug sites.

DISCUSSION

The 2003 fires provided a unique opportunity to increase our understanding of the way in which wildfire and its aftermath can impact on fish fauna. The

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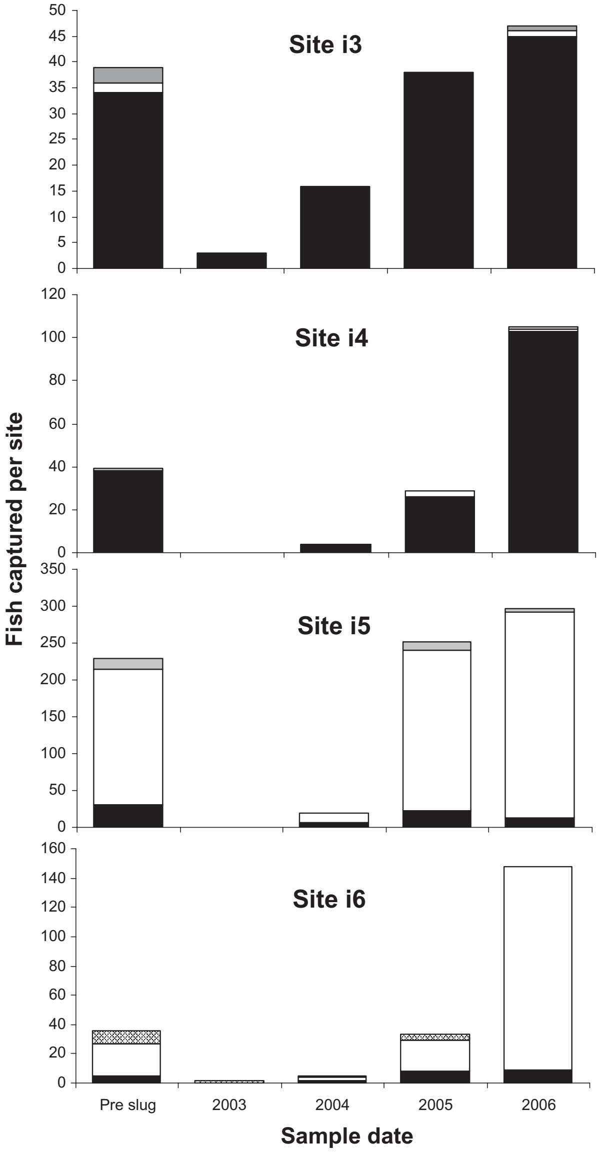

Fig. 7. Distribution and abundance of fish between five sampling occasions at the four most impacted sites. Solid bars = *G. bispinosus*; Clear bars = *G. olidus*, Grey bars = salmonids; Hatched bars = other species. Note differences in *y*-axis (total catch).

study was opportunistic but indicated that the post-fire sediment slug in the Buckland and Ovens rivers almost totally extirpated fish from the higher altitude sites in this catchment. Fish abundances were reduced by 90–100% from the source of the sediment slug at Dingo Creek (520 m above sea level (masl)) and downstream to the area around the township of Ovens (220 masl), a total distance of approximately 55 km. While it is possible that some fish escaped by moving downstream or into refuge tributaries it appeared that a considerable decrease in fish abundance was due to direct mortality. Evidence for this came from observations of dead fish during and immediately after the sediment slug and the lack of large increases in fish abundance at downstream sites. A number of previous studies have also observed 100% reductions of fish populations following post-fire sediment slugs (Novak & White 1990; Rinne 1996; Riemann *et al.* 1997).

During and immediately after the sediment slug, dead and stressed fish were found as far as 80 km downstream of the source, however, post-sediment slug sampling did not indicate any discernible impacts of the sediment slug on fish abundances below 55 km downstream. While no decreases in fish abundance were measured at the low altitude sites they were, however, subjected to decreases in water quality and increased sedimentation but at a reduced level to upstream areas. It appeared, therefore, that while some fish may have died in the lower altitude areas, the magnitude of these reductions was not great enough to be measured by our sampling regime. Rieman and Clayton (1997) also documented a decrease in impacts from low-order streams to high-order streams as a result of dilution as high suspended sediment concentrations descend down the catchment as a result of post-fire sediment slugs.

Impacts during and immediately after the sediment slug included observations of dead fish, stressed fish gulping at the water surface and freshwater crayfish walking out of the stream. While water EC increased, it did not exceed 200 μs cm^{-1} which is within the tolerance limits of all the fish species normally inhabiting the Ovens River. In comparison, turbidity and dissolved oxygen had deteriorated to levels that were detrimental to fish and this was likely responsible for the impacts on fish and freshwater crayfish. In previous studies increased suspended sediment has also been associated with dead fish following post-fire sediment slugs (Bozek & Young 1994; Rinne 1996). Furthermore, McKinnon (1995) observed freshwater crayfish walking out of Broken Creek, in south-eastern Australia, a phenomenon attributed to low-dissolved oxygen.

Deposited sediment decreased the availability of habitat in the impacted areas due to a smothering effect. These impacted areas normally consisted of cobble and boulder substrate interspersed with some wood debris (authors' observations 1999). It is unlikely that recolonization of these areas would have occurred while this sediment was present due to a lack of structural habitat. Spawning of the mountain galaxias and the two-spined blackfish in these areas may also have been reduced by a lack of appropriate spawning habitat. Both of these species deposit their eggs in gravel/cobble substrates (Cowden 1988; O'Connor & Koehn 1991; O'Connor & Zampatti 2006), a habitat type that was noticeably absent as a result of the deposited sediment. Indeed, the results of the current study indicated that spawning of these fish was reduced in impacted areas until the 2005 spawning

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season. Moreover, relatively few young-of-year fish were sampled in 2004 relative to the large number of young-of-year fish that were sampled in the following year, when sediment had been washed downstream. Previous studies have indicated that large amounts of deposited sediment may interfere with spawning in the two-spined blackfish (Doeg & Koehn 1994).

Two-spined blackfish are not generally regarded as migratory and have a home range limited to less than 20 m (Lintermans 1998). Furthermore, while a propensity for some movement has previously been described (Koehn 1987), the distances involved in that study were potentially limited as fish were merely observed colonizing a short reach of 'rehabilitated' river. Our data suggest a revision of this movement model with two-spined blackfish recolonizing an entire reach of river many tens of kilometres long. Some areas in this reach were at least 5 km from the nearest feeder stream. The nature of this recolonizing mechanism is unknown but may rely upon the uninterrupted movement of fish into uninhabited areas where there is no competition with individuals of either their own or other species (i.e. introduced trout). This is supported by Lintermans (1998), who discussed the possibility of 'stray' two-spined blackfish, which could be important in recovery after an environmental disturbance. Salmonids have also been shown to recolonize streams quickly after similar sediment slug events in North America (Novak & White 1990). For these reasons, it could be expected that trout might be the first species to recolonize these areas after such a disturbance. However, we found that *G. bispinosus* was the first species to recolonize all badly affected sites. Rieman and Clayton (1997) described the importance of refugia (such as tributaries and side-channels) during such events. Such areas allow for transfer of fishes back to the affected area after the disturbance has passed.

The results of this study indicated that there was a major decrease in fish abundance in the upland impact sites as a result of the post-fire sediment slug. However, there are some limitations associated with this study, in particular the reduced amount of data available from control sites from the immediate post-fire sample. However, while these data were not as extensive as other yearly control data, the impacts on the upland impact sites during this period were so extensive that the impacts of the sediment slug were still obvious despite this limited amount of control data.

Previously, the proximity and relative location of refugia, unimpeded routes of access and the occurrence of complex life history patterns have been recognized as influencing recolonization of fishes (Gresswell 1999). Indeed, the movement of fish into affected reaches of river from refuges many kilometres away reiterates the importance of maintaining connectivity of habitats in the recovery of fish populations after decimation from natural events such as fire. Previous studies have also concluded that fish populations can recover quickly (within a few years) following large disturbances (Rieman & Clayton 1997), and it has been suggested that the ability to do this is probably linked to adaptation of these species to periodic disturbances such as fire (Gresswell 1999).

The Buckland River has several small tributaries along its course, which, while burnt during the fire, were not affected by sediment slugs, and could have acted as a source for fish to recolonize the main channel. Many species of native Australian fishes, including the blackfish and percichthyids, fan their eggs during incubation. This reflects the often high sediment loads of Australian waters, even in upland areas. During times of increased sediment loads, such as after a mass wasting event, these behaviours could mean the difference between successful recruitment and the failure of a cohort. The relatively large number of young-of-year blackfish sampled in the high-impact areas in 2005 (24 months post fire) indicates that successful spawning had occurred in the previous months. In contrast, trout eggs, which are generally laid in gravel and then left to hatch, are at risk of being smothered by sediment in such a situation. There are many examples of massive decreases in hatch success for trout and salmon when high sediment loads covered redds (Cordone & Kelley 1961; Newcombe & MacDonald 1991). It has also been reported that the chronic impact of fine sediment accumulation in substrates may have as great an influence on salmonid populations as the initial sediment and ash input (Rinne 1996).

The potential impact of post-fire changes on small, isolated fish populations can be devastating (Spina & Tormey 2000; Dunham *et al.* 2003). An interesting case study in south-eastern Australia is the endangered barred galaxias (*Galaxias fuscus*), which due to an isolated range is at risk of being significantly impacted by fire. The barred galaxias is only found in small streams in the upper reaches of the Goulburn River catchment of northern Victoria. These populations are at considerable risk if fire were to burn through parts of the range of this species. Potential extirpation of threatened and/or isolated fish populations has previously been canvassed. Gresswell (1999) noted that in some watersheds, populations of native fish have become increasingly isolated in headwater streams because of land use practises and non-native introductions and in such cases the reduced habitat connectivity has increased the vulnerability these populations should be to disturbance such as fire. It is suggested that such populations are included on high-priority asset registers for fire controllers to consider when planning fire strategies.

While this study was undertaken solely within the Ovens River catchment, given that the fires burnt a much greater area encompassing over 1.3 million hect-

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ares in Victoria alone, then the impacts measured from the study area could also be applied across a much broader landscape. Furthermore, the removal of dams and weirs to enable fish passage has been considered as a potential solution to barriers to fish movement at redundant structures. However, the removal of these structures could also mobilize large amounts of entrapped sediments which could subsequently be washed into downstream habitats where they could have potentially detrimental impacts similar to the current study. Indeed, Doeg and Koehn (1994) found the impact of desilting a small weir in south eastern Australia introduced large amounts of sediment into the downstream areas of the creek, resulting in decreases in native fish abundances.

The unique nature of this event, and our ability to opportunistically measure recovery of what was, in essence, a localized extinction of all species, provided valuable information both for fish biologists and waterway managers. The current study has indicated that fish populations that are residing in well-connected streams with diverse habitat types are capable of surviving and recolonizing after catastrophic natural events. Consideration needs also be given to the impact of such events on isolated and threatened fish populations that may not be able to recover using the mechanisms described here.

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