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Impact of Timber Harvesting and Production on Streams: a Review

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Abstract

Timber harvesting operations have significant effects on both water quantity and water quality. The effects on water quantity have been well documented both in Australia and elsewhere. The effects on water quality are less widely appreciated, and include elevated concentrations of dissolved salts, suspended solids and nutrients, especially during peak flow periods. Several Australian studies have failed to measure peak flow transport of suspended solids, or have measured it inadequately, thus severely underestimating transport.

The major short-term effects of timber harvesting on the aquatic biota result from increased sediment input into streams or increased light through damage to, or removal of, the riparian vegetation. Sediment which settles on, or penetrates into, the stream bed is of more concern than suspended sediment, and can lead to long-term deleterious changes to fish and invertebrate populations. Increased light causes an increase in stream primary production which may increase invertebrate densities, and alter community composition. These biological consequences have not yet been adequately investigated in Australia. Longer-term effects, as yet not investigated in Australia, include changes to stream structure as the regrowth forest has fewer large logs to fall into the stream. These large logs play a major role as habitat and retention structures in streams.

There has been no attempt to evaluate the effects of timber production activities, including pesticide use and fuel reduction burning, on the Australian stream biota. Likewise, although buffer zones are widely advocated as a protection measure for streams in Australia, there have been no studies to evaluate their effectiveness.

Introduction

The headwaters of most of Australia's rivers occur in forest and, because impact on the headwaters can affect the downstream reaches, the potential influence of the forestry industry on Australian streams is large. Much of the water destined for off-stream uses (e.g. domestic and agricultural water supply) and in-stream uses (e.g. maintenance of fisheries and aquatic ecosystems) is ultimately derived from areas where logging is in progress or is planned and, although there is now widespread appreciation of the potential impacts of forest harvesting on water yields, there is far less appreciation of the potential impacts on water quality, especially biological water quality. This review is intended to provide a critical summary of the literature on the impacts of forestry operations on streams, emphasizing Australian data and, particularly, impacts on biological water quality.

Most of the available literature deals with the effects of timber harvesting activities including establishing access to the timber, falling and removal of trees and establishment of regrowth. The impact of these activities on stream environments constitutes much of this review. A number of other activities, such as pest and fire control and conversion of native

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forest to exotic plantations, may also be applied in timber-production operations when specific conditions necessitate their use. Since there is far less information available on the impact of these activities, they are discussed separately.

Effects on Stream Flow

The effect of forest harvesting operations on stream flows is probably the most widely and intensively studied environmental aspect of forestry practice (Hibbert 1967; Bosch and Hewlett 1982). The influence of catchment vegetation particularly forests, on stream discharge depends upon the relative importance of a large number of variables, many of which are quite site specific. As a result, while some general trends can be determined from the literature, the effects of any particular alteration to catchment vegetation often cannot be predicted with great confidence.

In general, removal of forest vegetation, by decreasing evapotranspiration, increases stream runoff whereas reforestation decreases it. The increase in stream flow following harvesting usually depends on the amount of timber removed, i.e. the intensity of the operation, and the proportion of the catchment harvested (e.g. Hewlett and Hibbert 1961; Reinhart and Eschner 1962; Rothacher 1971; Ffolliot and Thorud 1975; Hornbeck 1975; Rich and Gottfried 1976). The increase in stream flow following harvesting gradually diminishes over the following years (Hibbert 1967; Rothacher 1971; Aubertin and Patric 1974), except in cases where the regrowth of vegetation is actively discouraged (e.g. Likens *et al.* 1970; Hornbeck 1975). Australian results have generally been similar (Gilmour 1971; Tsykin and Laurenson 1980). In the Coranderrk catchment (Anon. 1980*a*, 1980*b*, 1980*c*), water yield in Picaninny Creek doubled from an average of 256 mm to a maximum of 564 mm in the year following the clearfelling of an old-growth mountain ash (*Eucalyptus regnans*) forest in the summer of 1971–72, but Picaninny Creek streamflow had declined to about 100 mm below pre-treatment levels by 1985 (P. O'Shaughnessy, personal communication).

Several other catchment studies of the relationship between forest harvesting and streamflow are under way in Australia but have yet to produce published results on the effects, although baseline data have been published (e.g. at Cropper Creek, Bren *et al.* 1979; Bren and Leitch 1986).

Effects on Water Quality

Suspended Solids

Water from rainfall enters the stream either as seepage flow through the soil mantle or as runoff over the soil surface (Packer 1967). Overland flow is rare in undisturbed temperate forest because the soil strata are usually able to absorb all the precipitation reaching the ground (Pierce 1967). The destruction of protective vegetation and compaction of the soil surface that is associated with timber harvesting procedures reduces soil permeability to water, increasing erosive surface runoff. The duplex soils, widespread in south-eastern Australia, are particularly prone to 'sealing' when exposed to rainfall (Walker 1986) and this combined with the high clay levels in Australian soils (Papadakis 1969) means that the risk of elevated stream turbidity resulting from activities such as timber harvesting is greater in Australia than for many other areas.

It is difficult methodologically to measure sediment or suspended solids transport in streams. Most of the sediment is transported during periods of high flow (Crickmay 1974), often with a large proportion of total annual load being transported in the three or four largest floods. Figure 1, for example, illustrates the concentration of suspended solids present in Magela Creek, Northern Territory, during a flood event. The combination of high concentration and high flows means that a large total load is carried in a very short time. Similar results, but with more complex hydrographs, have been published for a variety of temperate streams (e.g. Paustian and Beschta 1979; Weston *et al.* 1983; Olive and Rieger

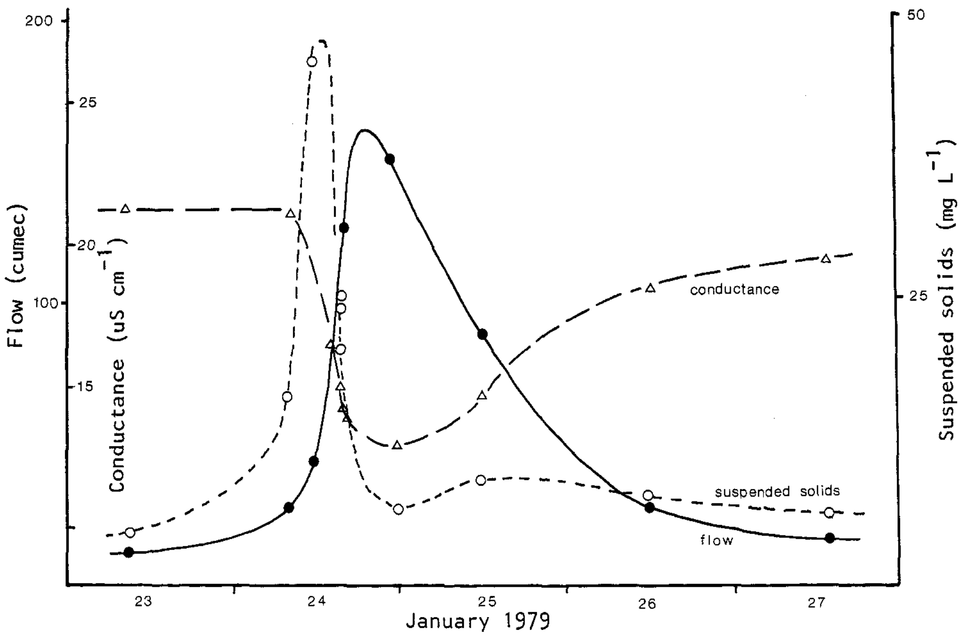

Fig. 1. Concentration of suspended solids, conductance and discharge during a flood event on Magela Creek (Northern Territory) demonstrating the classical relationship between discharge and concentration of suspended solids (from Hart *et al.* 1982).

1987). The concentrations of suspended sediment in Fig. 1 are very different on the rising and falling limbs of the hydrograph. Similar behaviour will also be exhibited by materials such as metals, some pesticides and nutrients such as phosphorus, that are transported adsorbed on to, or associated with, particulates (e.g. see Hart 1986). The concentrations of any of these materials plotted against instantaneous discharge in a time series for a particular flood event results in a hysteresis curve. Such a curve for the flood event depicted in Fig. 1 is shown in Fig. 2. The form of the hysteresis curve for any particular flood event

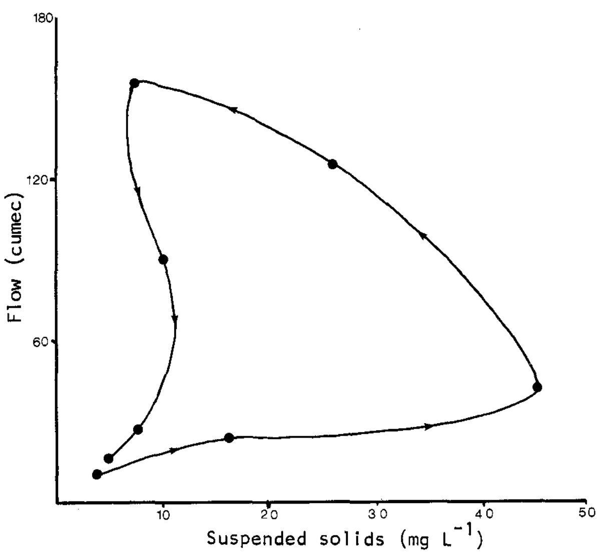

Fig. 2. The hysteresis curve obtained when the data for suspended solids concentration and discharge presented in Fig. 1 are plotted in a time series.

varies depending on the amount of rainfall, where and when the rainfall occurs in different parts of the catchment, the length of time since the last flood event and the size of that event and so on. Olive and Rieger (1985) have illustrated the variability of hysteresis curves for sediment concentration during storm events for small catchments in south-eastern New South Wales.

The plotting of concentrations of suspended solids for a series of weekly, or less frequent, samples against discharge at the time of sampling simply takes a more-or-less random selection of points from a number of different hysteresis curves and should not be expected to produce a simple relationship. Both Feller (1981) and Bren *et al.* (1979) failed to find statistically significant simple regression relationships between nutrient or particulate concentrations and instantaneous flow in small streams. This result, which would be expected for the reasons above, does not support their contention that concentrations found from weekly water samples multiplied by weekly discharge values therefore give accurate nutrient or sediment load results. Weekly sampling is likely to miss the rising stages of flood events and therefore underestimate loads. The only way to test whether this is the case is to sample through flood events. The plotting of sediment concentration against instantaneous discharge has also been used erroneously by various authors either to demonstrate that there is no relationship between discharge and suspended solids concentration, or to produce a 'rating curve' which can then be used to calculate loads of suspended solids transported by streams from discharge data. Walling (1977) and Rieger *et al.* (1982) both found the rating-curve method inadequate.

In spite of these problems, many recent Australian studies have been based on samples collected weekly or less frequently, with loads being calculated by multiplying discharge by concentration or from the statistically invalid regression relationship between load (i.e. concentration × discharge) and instantaneous discharge (e.g. Hopmans *et al.* 1987). Olive and Rieger (1987) compared estimates of the transport of suspended solids for a series of flood events in two streams in southern New South Wales calculated by the rating-curve method and from actual measurements. They found that the rating-curve method underestimated the load by between 73 and 82%. Because of these errors, most early, and some recent, estimates of sediment loads are substantial underestimates.

An increase in sediment accession to the stream during and following logging procedures has been well documented. Early studies include those of Lieberman and Hoover (1948) in the United States of America, and there is now a considerable body of data from the USA (e.g. Hornbeck and Reinhart 1964; Haupt and Kidd 1965; Packer 1967; Brown and Krygier 1971; Burns 1972; Aubertin and Patric 1974; Beschta 1979; Gurtz *et al.* 1980; Nolan and Janda 1981), Canada (e.g. Culp and Davies 1983) and elsewhere (e.g. Graynoth 1979).

The major sources of the increased sediment appear to be roading (including snig tracks and stream crossings) and land slips (Beschta 1979; Olive and Walker 1982). Lieberman and Hoover (1948), Hornbeck and Reinhart (1964), Fredrikson (1965), Haupt and Kidd (1965), Brown and Krygier (1971), Burns (1972) and Reid and Dunne (1984) all noted that roading was the major source of increased sediment loads from harvested catchments. In several cases (e.g. Fredrikson 1965; Brown and Krygier 1971) the increase in sedimentation was noted in the period between road construction and the commencement of falling and timber removal operations. In other cases, when trees were felled but not removed from the catchment, (i.e. no roads or snig tracks were constructed) no increase, or only a very small increase, in sediment runoff was found (Lieberman and Hoover 1948; Likens *et al.* 1970). Landslips are more frequent in logged areas (Swanston 1971, 1981) and may add significantly to the sediment loss from the catchment following timber harvesting (Fredrikson 1971; Burns 1972; Kelsey *et al.* 1981; Rice and Datzman 1981).

The amount of sediment lost from a catchment depends on site factors such as slope and soil type (e.g. Brown 1983) and also on the intensity of the harvesting operation, partly because of the higher density of roading and partly because of the more intensive site-preparation activities following harvesting (Brown 1983; Blackburn *et al.* 1986). Sullivan

(1986) failed to find large-scale long-term impacts of forest harvesting on suspended-sediment loads after comparing differences in concentrations upstream and downstream of a large harvesting operation in Oregon, but the presence of large-scale intensive forestry operations upstream of her upper site make the results difficult to interpret.

Results from Australian studies have been consistent with those obtained elsewhere, with increased sediment runoff following logging, particularly on the rising limb of the flood hydrograph, and road crossings often being identified as primary sources of sediment (Gilmour 1971; Olive *et al.* 1979; Anon. 1980*a*, 1980*c*; Burgess *et al.* 1981; Gilmour *et al.* 1982; Borg *et al.* 1987).

Cornish (1980, 1981, 1983) investigated turbidity levels in a number of streams draining catchments from which the timber was harvested in New South Wales. In each of his studies, he recorded higher maximum turbidities from streams with logged catchments than for similar streams with unlogged catchments, and in all but one study the mean turbidities were higher in the logged than the unlogged streams. Nevertheless he concluded that 'in general, forest operations do not have an adverse impact on stream turbidity levels', which is inconsistent with his results.

Olive and Rieger (1987) found elevated levels of transport of suspended solids in logged catchments in southern New South Wales, but noted that the timing and intensity of rainfall events following harvesting operations has a major influence on the amount of sediment which enters the stream. They note that sediment transport in the stream tends to be source limited.

Deposited Sediment

Most investigations of stream sedimentation resulting from timber harvesting operations have concentrated on suspended sediment. However, as noted above, high concentrations of suspended sediment are present in streams primarily during relatively brief periods of high flow. Sediment deposited on the stream bed is likely to be of far greater significance to the stream biota, for reasons which will be discussed below. Unfortunately, there are far fewer data available on the extent of the phenomenon.

Tebo (1955) and Lemly (1982) both noted increased bedloads of sediment in streams draining logged catchments. Moring (1982) attributed a significant decrease in gravel permeability following intensive clearfelling operations in an Oregon catchment to the penetration of additional fine sediment into the stream bed. Brown and Krygier (1971) and Golladay *et al.* (1987) have both pointed out that, even after sediment input to the stream has returned to normal as a result of forest regrowth, redistribution and transport of deposited sediment within the stream may continue for many years, continuing to disturb the instream communities.

The deposition of sediment in streams resulting from timber harvesting activities has not yet been investigated in Australia, although the presence of large quantities of deposited sediment resulting from other catchment activities which produce increases in stream sediment levels has been noted (Davey *et al.* 1982, 1987; West *et al.* 1984). Deposited sediment from the construction of the Thomson dam penetrated the interstices between the stones, filling them up to a depth of at least 60 cm (Davey *et al.* 1987), a phenomenon previously demonstrated in the laboratory by Beschta and Jackson (1979). While deposited surface sediment may be quickly flushed out of the system by ensuing spates, major floods are probably necessary to flush out the fine sediment trapped within the stream bed.

Nutrients and Dissolved Solids

Streams draining natural, undisturbed forests in Australia generally have very low nutrient levels (Campbell 1975), reflecting the generally low nutrient status of Australian soils (e.g. Wild 1958; Beadle 1962) and the ability of forest systems to trap and retain nutrients (Likens *et al.* 1977; Bormann and Likens 1979). Catchments which have been disturbed by human activity tend to 'leak' nutrients, and the levels in the streams draining them tend to be higher (e.g. Campbell 1978).

Following timber harvesting, the amount of dissolved nutrients leached from organic debris into the soil and thence to the stream via subsurface flows increases, while additional nutrients may also be adsorbed on to inorganic particulate material or contained within organic particulate material which is washed into the stream via overland flows (Fredricksen 1971). Of the two, the first process is not likely to be of great significance except in very sandy soils since nutrients, and phosphorus in particular, tend to be rapidly and tightly bound to soils with a significant clay or organic content (Cullen *et al.* 1978).

Invariably, significant increases in dissolved nutrients have been reported when catchments have been deforested or harvested. Likens *et al.* (1970) reported some of the largest increases in most major ions following deforestation of a catchment in the Hubbard Brook Experimental Forest, New Hampshire. Nitrate concentrations were 41-fold higher during the first year, and 56-fold higher during the second, than in a similar undisturbed catchment. However, in this study fallen timber was not removed from the catchment and regrowth was actively discouraged by the use of herbicides, thus increasing the amount of material available for decomposition on the catchment slopes and reducing the subsequent uptake to nutrients by the active regrowth.

Studies involving commercial-style harvesting have also found increases in dissolved nutrient concentrations, although smaller than those found by Likens *et al.* Fredricksen (1971) in Oregon, Aubertin and Patric (1974) in West Virginia and Webster and Waide (1982) in North Carolina all noted nutrient levels in streams elevated several-fold following harvesting of the catchment.

Although the actual concentrations of nutrients in the stream are the most significant factor in influencing the instream community in the immediate area of the harvested forest, it is the total nutrient load which is most significant for downstream uses such as reservoirs (e.g. Cullen *et al.* 1978). There is an extensive literature on the effects of such loads, reviewed in the Australian context by Wood (1975).

Webster and Swank (1985) noted that concentrations of nutrients bound to suspended particulate material in streams are higher than the concentrations of dissolved nutrients. The amount of particulate transport of nutrients in streams seems to a large extent to be determined in undisturbed streams by instream processes, including biological activity and presence of retention structures such as debris dams. Thus, the increase in particulate nutrient transport which is found after timber harvesting may be due both to increased accession of particulate material to the stream, as noted previously, and to disturbance to the biota and instream processes discussed below.

Concentrations of dissolved solids other than nutrients also increase in streams following catchment harvesting. This may be due to increased leaching of salts from the soils or to the raising of more saline groundwater levels by increased percolation of precipitation. Increases in ionic concentrations have been noted in New Zealand (Graynoth 1979), the USA (Likens *et al.* 1970; Webster and Waide 1982) and Australia (Anon. 1980*a*, 1980*b*). However, no detectable change was observed in the concentration of dissolved solids following selective cutting in Blue Jacket Creek in Victoria, and Cornish (1980, 1981) reported that logging in New South Wales had no significant effect on electrical conductivity values of the streams he studied. In other areas, particularly in Western Australia, increases in stream salinity following catchment clearance and timber harvesting are now recognized as major constraints in catchment management (Shea *et al.* 1975, 1978).

Dissolved Oxygen

Dissolved oxygen in turbulent upland streams is nearly always in saturation equilibrium with the air (Hynes 1970). However, the decomposition of organic material requires oxygen, and the combination of large amounts of logging debris and periods of low flow may produce significant reductions in levels of dissolved oxygen in streams (Eschner and Larmoyeux 1963; Likens *et al.* 1970). Burns (1972) recorded a large drop in dissolved oxygen due to decaying slash material in pools associated with road construction near a stream.

Penetration of finely divided organic debris, or organic material adsorbed to fine sediment, into the stream bed following catchment harvesting can result in substantial decreases in interstitial oxygen concentrations (Hall and Lantz 1969). This may be attributed partly to the oxygen demand of the organic material and partly to the lower intra-gravel water flow. The effects may persist for 6 years or more, supporting the idea that subsurface material is not easily removed. In Australia, a decrease in stream oxygen concentration was not observed under carefully controlled logging conditions (Anon. 1980*b*).

Organic Material

Terrestrial organic material entering a stream from the catchment plays a critical role in stream ecology (Cummins 1986). It provides both a source of energy and a physical habitat for the stream biota; thus, changes to the amount and types of organic input to the stream have significant ecological impacts. Large items of organic debris, such as trees, that fall into streams contribute to stream structure, helping to form the pools which provide fish habitat. Leaves and other small items of debris form a food source for many stream invertebrates. Two sets of impacts may result from forest harvesting, one from the deposition of excess logging debris in the stream, and the other from changes to the nature of the forest resulting from harvesting operation.

Although few quantitative data are available, several authors have reported large amounts of organic debris deposited in streams as a result of logging activities (Narver 1971; Beschta 1979; Graynoth 1979), and this phenomenon is common in Australia, as illustrated by Campbell (1986). The need to remove such debris to protect aquatic systems is now widely recognized (Froelich 1971; Burns 1972; Ponce 1974). However, even where large organic debris, such as branches, is removed from the stream, large amounts of finely divided material such as needles, leaves and broken twigs often remain (Ponce 1974; Gurtz *et al.* 1980). Programmes for the removal of logging debris may also cause problems if they remove logs which were naturally present before the harvesting operation, since this may reduce habitat for fish (Elliott 1986).

As the forest regrows following logging, there are several consequences for the organic material entering the stream. Initially, there will be a decrease in the amount of leaf material entering the stream (Webster and Waide 1982) and, since regrowth forest is different in composition from mature forest, there will also be a change in the types of leaf material (Webster *et al.* 1987). In Australia, *Acacia* species tend to be abundant in regenerating forest, and the only information available for *Acacia* leaves indicates that *A. melanoxylon* leaves are not palatable to stream invertebrates (O'Keefe 1982; Campbell, unpublished data).

The rate of breakdown of leaf material entering the stream may be elevated in harvested catchments (Webster and Waide 1982). This may result from increased nutrient concentrations in the water, since these are known to increase leaf breakdown rates (Elwood *et al.* 1983) perhaps by promoting microbial growth.

At some stage following logging, the amount of large organic material within the stream declines as the source of input is removed, because of the absence of mature trees in the regrowth forest (Webster and Waide 1982; Culp and Davies 1983). This reduction may persist for many years following harvesting activity. Silsbee and Larson (1983) found lower levels of very coarse debris, such as logs, in streams in logged catchments, compared with unlogged ones, 45 years after logging was discontinued. These logs help form the debris dams which are important retention structures in natural streams, reducing the rate of nutrient loss from the catchment, retaining particulate material and providing habitat for the stream biota (Bilby and Likens 1980). Webster and Swank (1985) found lower numbers of debris dams in logged compared with undisturbed catchments at Coweeta, and Golladay *et al.* (1987) suggest that the numbers of debris dams may be depressed for 100–400 years in streams in logged catchments.

Light Availability and Temperature

The effects of timber extraction on light availability in streams has received little attention until recently. Newbold *et al.* (1980) found that angular canopy density over northern Californian streams was about 80% in unlogged catchments and 20% in logged catchments. Where buffer strips were retained, densities varied from 20 to 100%. Culp and Davies (1983) also found that light availability was increased in a stream in British Columbia as a result of logging. Murphy *et al.* (1981) concluded that the major ecological effects of logging on small streams in the Oregon Cascade Range were a result of increased light availability, and that they were so great that they masked effects of increased sedimentation.

The only Australian data on the effects of riparian vegetation on light availability in streams are that of Metzeling (1977), who found an increase of more than an order of magnitude in the amount of light reaching the surface of the cleared versus forested sections of two Victorian streams.

Streams shaded by large stands of bankside vegetation generally have lower summer and higher winter temperatures and a smaller daily range of temperatures when compared with similar unshaded streams. Hence, one could expect to observe a change in the stream temperature regimes after timber harvesting operations that include the removal of trees and shrubs up to and including the stream bank (Eschner and Larmoyeux 1963; Gray and Edington 1969; Likens *et al.* 1970; Burns 1972; Graynoth 1979; Webster and Waide 1982; Culp and Davies 1983).

Changes in stream temperature regime are smaller or absent where stream buffer strips are retained (Brown and Krygier 1970; Brown 1971). Brown (1971) notes that, to avoid temperature changes, buffer strips must be provided along the entire length of stream. Because the rate of heat loss at the stream surface is smaller than the rate of heat accession, the provision of short sections of shaded stream will not cancel the effect of similar lengths of exposed stream. Burns (1972), for example, recorded rates of increase in water temperature of 1·0°C 100 m^{-1} in blocks cut to the streambank, but decreases of only 0·5°C 100 m^{-1} in downstream uncleared blocks.

Metzeling (1977) found that the cleared sections of two streams in the catchment of the Yarra River displayed higher temperature ranges as well as higher summer temperatures than partly cleared or forested sections of the same streams. However, the Melbourne Metropolitan Board of Works (Anon. 1980*b*) were unable to detect changes in temperature as a result of clearfelling the catchment of Picaninny Creek, where a 20-m buffer strip was provided.

Effects on the Stream Biota

There is far less information on the impacts of timber harvesting activities on the biota of streams than on their physical and chemical characteristics. Where biological studies have occurred, particular emphasis has been placed on fish, especially salmonids, because of their recreational and commercial value. Few studies have considered the overall biological impact, or the impact on the instream flora and invertebrate fauna.

Long-term biological effects arise mostly as a result of alteration to the stream riparian vegetation, which is intimately connected with the instream biota (Cummins 1986). Shorter-term effects are mostly attributable to the impact of suspended or deposited sediment. Boschung and O'Neil (1981) have published a study which failed to find an effect on stream fish and invertebrates following forest harvesting in Alabama; however, less than 12% of the catchment was harvested, and it is unclear from the paper whether this area was near the stream.

Effects on the Periphyton

Data on the effects of forest harvesting on the periphyton, or attached algal communities, in streams are inconclusive. Lyford and Gregory (1975), Graynoth (1979), Murphy *et al.* (1981) and Shortreed and Stockner (1982) all observed generally increased biomass of periphyton in streams with harvested catchments. Filamentous algae in particular tend to become more common where the stream is unshaded. However, neither Culp and Davies (1983) in British Columbia nor Winterbourn (1986) in New Zealand found significantly higher periphyton biomass in streams with logged versus those with unlogged catchments. Although, as Winterbourn (1986) points out, this does not necessarily mean that primary production levels were the same. Similarly, although algal blooms did occur at three of the logged sites in the study by Newbold *et al.* (1980) there was no consistent pattern in the primary production levels in the streams. The inconsistent results in these studies could be partly due to low nutrient levels limiting periphyton growth in some streams.

Effect on Macroinvertebrates

It is clear from several North American studies that the changes in stream macroinvertebrate communities caused by forest harvesting may be long term. Erman *et al.* (1977), Haefner and Wallace (1981) and Silsbee and Larson (1983) all found changes in stream invertebrate faunas, which they attributed to logging activities, many years after logging had ceased. In the last case, logging had ended 40 years before the study.

The major effects appear to arise from increased sediment suspended in the water and deposited on and in the stream bed. Suspended sediment may be expected to have its greatest effect on filter-feeding invertebrates. High concentrations of inorganic particles in the water may clog the nets and other filtering mechanisms by which these organisms feed. The densities of filter-feeding invertebrates are often reduced in streams receiving effluents containing high concentrations of suspended solids (e.g. Gammon 1970; Nuttall and Bielby 1973; Mayack and Waterhouse 1983). A similar effect was found in a North Carolina stream with a logged catchment (Lemly 1982).

Some years after the harvesting of a catchment, the abundance of filter feeders may be higher than in streams with undisturbed catchments (Haefner and Wallace 1981; Silsbee and Larson 1983). This probably results from increased quantity and quality of the suspended fine particulate organic material (FPOM) that these organisms utilize for food. Increased leaf decomposition rates, increased algal production and elevated nutrient levels would all be contributing factors. The quality of the FPOM and suspended particulate material, rather than particle quantity, is the factor limiting population densities of filter-feeding invertebrates in streams (Benke and Wallace 1980; Georgian and Wallace 1981; Haefner and Wallace 1981).

Elevated levels of suspended inorganic material in streams also increases invertebrate drift, which is a primary response of invertebrates to the onset of stressful conditions (Wiederholm 1984). This response has been repeatedly demonstrated in experimental studies (e.g. Gammon 1970; White and Gammon 1977; Rosenberg and Wiens 1978; Culp and Davies 1983). In an Australian study, Richardson (1985) found significant correlations between the drift rates of some taxa and the increase in turbidity produced by a logging road crossing a stream in a catchment near Bega.

Suspended sediment may also contribute to the scouring of organisms from their streambed habitat during times of high flow (e.g. Chutter 1969). Tebo (1955) found that the faunal reduction caused by floods in sections of a stream affected by logging was greater than the reduction in upstream areas unaffected by the logging. He attributed this difference to an increase in drift rates caused by the suspended sediment, and physical detachment of animals by the saltating bed load.

Inorganic suspended solids are of concern in streams immediately downstream of discharge points or during periods of high flow. Longer term impacts arise when the material settles on or into the stream bed. Far more attention has been focussed on the biological impact of sedimentation on the surface of the stream than penetration of sediment into the bed.

Any permanent siltation damages the invertebrate habitat in streams (Hynes 1973). The deleterious effects of stream sedimentation caused by forestry activities to invertebrate communities has been demonstrated in North America (e.g. Tebo 1955; Newbold 1977; Newbold *et al.* 1980; Lemly 1982; Culp and Davies 1983; Silsbee and Larson 1983), New Zealand (Graynoth 1979) and Australia (Robinson 1977; Richardson 1985).

The changes in stream communities resulting from stream sedimentation most widely reported are reductions in species diversity, reductions in biomass and changes in species composition. Whereas taxa requiring solid surfaces are often reduced in numbers, taxa such as oligochaetes and chironomid midge larvae, which are capable of utilizing fine particulate sediments as a habitat and food source, become more abundant.

Two Australian studies have examined the effects of forest harvesting operations on stream invertebrate faunas. Robinson (1977) sampled the macroinvertebrate communities in the streams in the Corranderk Experimental catchments (Anon. 1980*a*, 1980*b*) on two occasions, late summer and winter. Picaninny Creek and Blue Jacket Creek, with harvested catchments, had higher percentage similarities to each other than either had to the control, Slip Creek. The density of benthic fauna in the clearfelled catchment (Picaninny Creek) was appreciably lower than in either of the other two creeks, particularly during the summer, while the control site had a higher diversity, but also higher numbers of oligochaetes, than the other two. However, the absence of preharvesting data in this study limits the conclusions that can be drawn, since it is not certain that the differences were due to the harvesting treatment alone.

Richardson (1985) found significant differences between invertebrate samples from upstream and downstream of logging influences in Mumbulla Creek, southern New South Wales. The differences were most marked in invertebrates collected from sand and shingle substrates, and were evident for at least 9 months after logging operations ceased. She attributed the effects to increased sedimentation.

The abundance of macroinvertebrates found deep within the substratum of a stream (the hyporheos) may equal or even exceed that found on the surface (e.g. Coleman and Hynes 1970; Williams and Hynes 1974; Morris and Brooker 1979). The hyporheos is also an important refuge for many macroinvertebrate taxa in times of environmental stress such as drought (Bishop 1973; Culp and Davies 1983), and may also be involved in diurnal vertical migration patterns (Campbell 1980) and seasonal migrations associated with particular stages of the life cycle (Hynes 1970). Yet the biological effects of sediment penetration into the substratum of the stream remain largely uninvestigated. Hynes (1973) predicted that the filling of crevices that serve as macroinvertebrate habitat will have deleterious effects on the macroinvertebrate fauna. Grenney and Porcella (1976) found that secondary production fell as sediment filled the crevices in the stream bed. Graynoth (1979) attributed part of the change he found in the invertebrate fauna of a stream with a clearfelled catchment to a reduction in dissolved oxygen concentrations in the interstitial waters of the stream bed due to the breakdown of finely divided organic debris.

Work at the Museum of Victoria (Blyth *et al.* 1984; Doeg 1984; Davey *et al.* 1982, 1987) indicates that increased stream sedimentation has been responsible for dramatic changes in the macroinvertebrate communities in rivers affected by reservoir construction. The density of many taxa has been severely reduced as sediment levels in the streams increased, while the abundances of a few taxa, known to be tolerant of adverse conditions, have been greatly enhanced. Davey *et al.* (1982) suggest that the penetration of sediment into the river bed may be a far more serious long-term problem than has previously been thought.

Effects on macroinvertebrate communities have also been attributed to changes in periphyton communities following timber harvesting. In contrast to most other studies, Winterbourn (1986) did not find significant differences in benthic invertebrate community composition or functional feeding groups between logged and unlogged streams in the Maimai experimental catchment; however, there was an increase in the utilization of periphyton as an invertebrate food source in the logged streams. Murphy *et al.* (1981) and Hawkins *et al.* (1982) found higher densities of benthic invertebrates at clear-cut and second-growth sites in Cascade Mountain streams, compared with old-growth sites. They attributed this effect to increased periphyton growth and increased leaf input from the deciduous regrowth forest.

As yet, there are no data to indicate an impact on macroinvertebrates due to a decrease in input of riparian litter material following timber harvesting. In part, this may be due to a masking effect due to increased periphyton production, and transport of particulate organic matter from upstream areas may be sufficient to support the shredders. Finally, it may simply reflect the generalist nature of the feeding mechanisms of stream invertebrates and our inability adequately to categorize them (Hawkins *et al.* 1982).

Effects on Fish

Forestry activities may have an impact on fish populations indirectly as a result of the impact on invertebrate communities which form the main food source for fish, or directly due to habitat modification or lethality (Schrivener and Anderson 1982).

In general, catchment clearing deleteriously affects fish populations. Adult fish survive quite high levels of suspended inorganic sediment (Cordone and Kelly 1961), although high levels may interfere with behaviour (Muncey *et al.* 1979; Berg 1982). However, sediment deposition, particularly through its influence on mortality of eggs and alevins in gravel-spawning species, is a more significant cause of declines in fish populations (Cordone and Kelly 1961; Gibbons and Salo 1974; Morgan and Graynoth 1978).

Fish populations may also be reduced in streams with harvested catchments as a result of debris deposited in the stream. Large-size logging debris prevents the free movement of migratory fish over the length of the stream, disturbs the spawning beds by debris movement in high flows, and changes the discharge patterns by forming dams and barricades (Narver 1971; Beschta 1979). Organic debris also influences fish populations by reducing dissolved oxygen (Hall and Lantz 1969; Narver 1971; Burns 1972; Graynoth 1979). Excessive removal of debris may also reduce fish and invertebrate populations (Elliott 1986) by reducing cover and habitat. Changes in water temperature associated with catchment harvesting may also be sufficient to affect fish populations (Lantz 1971).

Wilzbach (1986) suggests that trout populations in streams with harvested catchments should initially increase but then decline in the long term. The factors which contribute to the short-term increase are an increase in the biomass of invertebrates due to increased algal growth in the unshaded stream, together with increased foraging efficiency by the fish due to increased light availability and reduced habitat complexity. In the long term, fish populations are reduced as the regrowth forest begins to shade the stream but does not provide the input of large organic debris which provides much of the habitat complexity of forest streams.

Where adequate buffer strips of intact riparian vegetation are maintained along streams, the impacts on fish populations may be greatly reduced (Burns 1972; Graynoth 1979).

In the only Australian study to investigate the relationship between fish populations and logging, Richardson (1985) found lower numbers of the common jollytail, *Galaxias maculatus*, in an area disturbed by logging road construction in New South Wales, compared with a similar habitat in an undisturbed adjacent catchment. The lack of pre-treatment data on the populations in the creeks preclude any conclusive evaluation of the result, but the observed increased sedimentation was likely to have effects on many aspects of the known life history of this species.

Timber Production Activities

There are four main activities carried out as part of forestry management operations in Australia which are not part of the direct timber harvesting activities but which may, nonetheless, affect streams. These are: the use of pesticides, the use of fertilizer, fire prevention activities including fuel reduction burns, and the replacement of native hardwoods with exotic softwoods. There is little direct evidence for the impact of any of these activities on streams but significant effects may be expected.

Pesticides

Insecticides and herbicides are the main two categories of pesticides used in forestry management. The use of both these groups of chemicals has been reviewed by Norris and Moore (1981). Insecticides are used irregularly over generally small areas to control outbreaks of insect pests such as phasmatids (e.g. Anon. 1983). However, the insecticides used are organophosphates, which are quite stable in slightly acid water [malathion, for example, has a half-life of 5 months in water of pH 6 (Hart 1974)], and are known to be acutely toxic to aquatic invertebrates at concentrations below $0 \cdot 1$ μg L^{-1}. Therefore, the impact on streams of spraying malathion should be monitored whenever spraying is carried out near streams.

Herbicides, such as 2,4,5-T, hexazinone and 3,6-DCPA are more commonly and widely used in forest management. They are likely to be less toxic to freshwater animals than the insecticides, but far more toxic to freshwater algae. The toxicity of many of these compounds to the Australian freshwater biota should be evaluated, especially for those materials for which there is no published information on aquatic toxicity. There is also a need to monitor the biological impact on streams of spraying operations. McKimm and Hopmans (1978) recorded a maximum level of 2,4,5-T in a stream of 10 μg L^{-1} after a spraying operation in Victoria, four times the maximum level of $2 \cdot 5$ μg L^{-1} recommended by Hart (1974) for the protection of aquatic ecosystems.

Fertilizers

The use of superphosphate, the major fertilizer used, is not intensive, and the risk to streams would appear to be slight unless they are small and unshaded.

Fire and Fire Management

Studies on the impact of bushfires on catchments have been reviewed by several authors (e.g. Langford and O'Shaughnessy 1977; Humphreys and Craig 1981; Clinnick 1984*a*, 1984*b*) but, in general, studies of the impact on streams have concentrated on water yield (e.g. McArthur 1964; McArthur and Cheney 1965; Brown 1972; Anon. 1980*a*, 1980*b*; Mackay *et al.* 1980; Lawrence 1981; Kuczera 1985) with far less information on water quality.

In general, wildfires cause increases in the accession of inorganic sediment, dissolved solids, nutrients and organic materials to streams [Brown 1972; McColl and Grigall 1975; Smalls (cited in Cullen *et al.* 1978); Burgess *et al.* 1981; Shea *et al.* 1981; Condina *et al.* 1984; Chessman 1986; Mackay and Robinson 1987; Olive and Rieger 1987]. The intensity of the effect depends on the nature of the catchment, the intensity of the fire, the period between the fire and the first significant rainfall event, and the intensity of that rainfall event.

Fuel reduction burns are often less intense than wildfires and their impact on streams may be expected to be less. Gilmour and Cheney (1968) found that a fuel reduction burn in a pine plantation in the Australian Capital Territory had little effect on soil water-infiltration rates, soil loss and soil organic content, but quite small changes in terms of the catchment may produce major changes in streams. Fuel reduction burns are usually conducted at times of the year when rainfall is greater and quite large areas are often burnt. In Victoria, for example, about 190 000 ha per year are burnt. The impact of roads, access tracks and firebreaks constructed for fire prevention would be similar to that of roads constructed for harvesting, so needs no further discussion here.

Plantations of Exotic Species

The primary activity of this type has been the establishment of plantations of *Pinus* species. These are generally established in areas that have been cleared for agriculture, cleared during timber harvesting or cleared of uncommercial native forest. Where existing forest is cleared, the effects outlined above in relation to timber harvesting will be encountered. The establishment of an exotic plantation may also result in other impacts. Firstly, young pines generally require the application of pesticides and fertilizers, which are not necessary for the regeneration of native forests. The potential impacts of these materials have been discussed above. Secondly, the change in the catchment vegetation will alter the nature, amount and timing of the input of terrestrial lead and other plant material into the stream and this may have a marked effect on the stream invertebrate community. O'Keefe (1982) demonstrated that densities of macroinvertebrate taxa that colonized the artificial litter bags were much lower at sites where the stream flowed through a pine plantation than where it flowed through a native eucalypt forest, presumably because of the lower palatability of the pine needles to the Australian fauna.

Observations of litter fall from Australian sclerophyll forests have indicated that maximum leaf fall from native eucalypts occurs during the summer months (Lake 1982). However, in pine forests the maximum leaf fall occurs during autumn and early spring (Bray and Gorham 1964), and the change in timing of detritus input into the streams may also have an effect on the native fauna.

Aquatic Protection Measures

In view of the deleterious changes to both the biotic and physicochemical characteristics of aquatic systems which may be associated with timber harvesting operations, it is surprising that there have been so few investigations into procedures that may ameliorate or eliminate these effects. Cameron and Henderon (1979) outlined many harvesting procedures which may minimize the environmental effects of harvesting operations, but there have been no published Australian studies attempting to evaluate their recommendations.

At present, aquatic systems are protected by series of forest harvesting prescriptions, sets of regulations prepared by State forestry departments (e.g. Anon. 1984). These prescriptions set limits on slope angles and coupe sizes and specify roading requirements and buffer zones. Although buffer zones along streams have been widely advocated to protect streams (e.g. Clinnick 1985) there have been no Australian studies to determine the effectiveness of, or appropriate widths for, buffer strips in forestry operations for the protection of aquatic communities. In North America, Newbold (1977) and Culp and Davies (1983) both evaluated the effectiveness of buffer strips for protecting stream invertebrate communities. Newbold suggested that buffer strips needed to be >30 m wide to be effective; Culp and Davies found that a 10-m wide buffer strip was not effective in protecting invertebrate communities.

Conclusions

The major impacts of forestry operations on the stream biota occur through sediment and debris deposition and alteration of the riparian vegetation. Many Australian soils are highly erodable and rich in dispersive clays; this, combined with the fact that the rainfall erosion index values for Australia are comparable to those elsewhere (McFarlane and Clinnick 1984), indicates that sedimentation of streams is likely to be at least as large a problem in Australia as elsewhere. This is confirmed by existing data, even though many Australian studies are badly flawed. The overwhelming majority of sediment transport occurs in streams during periods of high flow, and studies which fail to sample intensively through such events, and analyse the data on an event basis, produce such gross underestimates of sediment load as to be almost worthless. Such studies should no longer be countenanced.

Deposited sediment probably has a far greater impact on stream invertebrates than has suspended sediment. It may penetrate deep into the stream bed eliminating, or at least severely modifying, the important hyporheic habitat. The predominance of clays, with their small particle size, in Australian soils may make this problem more acute than elsewhere. Where the hyporheic habitat is modified, the impact may be long term, requiring a major flood to flush out the sediment.

Riparian vegetation contributes large woody debris, which plays an important role in physically structuring streams, and finer material such as leaves, flowers and twigs, which is used as a food source by many stream invertebrates. Deposition of excessive organic debris in streams may fill pools, create barriers and lead to oxygen depletion. Removal of large trees from the riparian zone will reduce inputs of large woody debris over the long term, and eventually lead to simplification of the stream habitat.

Where catchment vegetation is harvested, regrowth vegetation, especially *Acacia* species, may not be palatable to stream invertebrates; this may lead to changes in the instream community. If exotic species are planted, there may be both a decrease in palatability and a change in the annual timing of the organic input to the stream.

Higher nutrient concentrations, higher water temperatures and increased light availability in streams may all result from timber harvesting operations. All three may alter metabolic processes in the stream, and the invertebrate and fish communities. There have now been a number of studies on the physico-chemical and hydrological impacts of forest harvesting in Australia, but there is only one published study of the biological impact (Richardson 1985). This is of great concern in view of the long-term changes to fish and invertebrate populations which have been demonstrated in North America. Even more significant is the complete absence of any studies on the effectiveness of buffer strips as a protective measure in maintaining the ecological integrity of streams in areas being harvested for timber. Given that buffer strips are the major protection technique for streams in Australia, this lack is extraordinary.

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