BACKGROUND DOCUMENT 6

MANAGEMENT ISSUES RELEVANT FOR BIODIVERSITY CONSERVATION IN FRESHWATER ECOSYSTEMS

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1. Summary

Prior to European settlement, Tasmania's forested catchments were a complex mosaic of vegetation types with an attendant diversity of aquatic habitats. Fires, floods and droughts were natural processes, which were probably partially modified by aboriginal practices. Aquatic biodiversity is very resilient to such changes, but poor land management can lead to dramatic, and irreversible changes to the physical nature of streams and wetlands, and the ecosystem processes on which the biodiversity depends.

In production landscapes, aquatic biodiversity can best be conserved by ensuring that management occurs at multiple scales to maintain the resilience of freshwater ecosystems. Forest practices in Tasmania, especially over the last 20 years, have succesfully established a network of riparian buffers on larger streams, and these are likely to have been effective in ameliorating the effects of forestry on in-stream biota. These could be improved still further by maintaining some lateral connections between riparian zones and upslope habitats during forest management cycles, and some initiatives are already in train to realise this. Simple extensions of the riparian buffer system are infeasible for small headwater streams, so a proportion of the headwater network within each CFEV catchment should be conserved within any one management cycle to provide refuge for in-stream biota and connectivity for semi-aquatic and riparian-dependent species to disperse over catchment boundaries.

There is a growing body of predominantly physical and chemical research suggesting that single, uniform prescriptions applied across vastly different habitats and regimes often do not meet their objectives. Promising developments in "risk-based" approaches are noted, and a general shift in thinking towards contextually appropriate variations in prescriptions should lead to more defensible, albeit complex, forest practices.

2. Biodiversity objectives for aquatic habitats

Accepting that all forestry activity will have some impact on biodiversity and its habitats, the overarching objective for managing aquatic biodiversity in forest systems should be to:

maintain the resilience of freshwater ecosystems at multiple spatial scales

By this it is intended that forest practices should aim to maintain freshwater ecosystems at the broad landscape scale through to the fine, local coupe scale so that their ecosystem values (including their biotic communities and biophysical characteristics) are are conserved in the face of cumulative impacts of forest management and related activities. A qualitative "vision" for conserving ecosystem function and biodiversity in production landscapes is provided by Fischer et al.'s (2006) "guiding principles":

> *Landscapes should include structurally characteristic patches of native vegetation, corridors and stepping stones between them, a structurally complex matrix, and buffers around sensitive areas. Management should maintain a diversity of species within and across functional groups. Highly focused management actions may be required to maintain keystone species and threatened species, and to control invasive species.*

I propose that the framework for catchment planning units and conservation values be provided, in the first instance, by the Conservation of Freshwater Ecosystem Values Project (CFEV) of the Tasmanian Department of Primary Industry and Water (2007). CFEV delineates large drainage basins within which are delineated sub-catchments thus providing spatial scales for regional and Planning Context Unit (PCU) scopes respectively[1]. Complementary, species-specific support to this objective is, of course, provided by a number of other instruments (e.g. threatened species legislation; see Section 8).

To achieve this, the following sub-objectives are proposed[2]:

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[1] IBRAs are largely delineated on terrestrial features and are likely to provide an inferior regional basis for planning freshwater biodiversity objective to that offered by CFEV.

[2] Cross-references to supporting Sections in this report areappended at the end of the italicised justification for each item.

1. Maintain aquatic habitats, including retaining habitat diversity, water quality and ecological flows so that their ecosystem values (including their biotic communities and biophysical characteristics) are maintained within the range of natural variation over time.

   Aquatic ecosystems are dynamic, and resilient to wide range of natural disturbances, so it is futile to aim for a "static" configuration of species. Instead, the landscape mosaic should include a range of forest types and ages so that aquatic species are able to persist somewhere within the landscape (Sections 6.3, 6.4, 7.2).

2. Manage riparian[3] zones so that ecosystem processes for aquatic systems are maintained or enhanced.

   Riparian zones provide the bulk of the energy base and important in-stream habitat elements for aquatic biota (Section 6.3). *Riparian protection of larger perennial streams has also been important in erosion control and reducing the inputs of sediment, nutrients and pesticides* (Section 5.3).

3. Conserve a proportion of Class 4 stream catchments within a CFEV sub-catchment during a harvest cycle.

   Because of the density of small headwater streams, extension of a formal streamside reserve network beyond that currently impelmented for other purposes (e.g. erosion control, safety, other steep terrain issues) is unlikely to be practical. Accordingly, it will be necessary to conserve a representative proportion of the headwater network within the harvest cycle of each sub-catchment to ensure the persistence of any aquatic or semi-aquatic species that specialize in headwater streams or need to disperse across catchment boundaries in order to persist (Sections 7.5 and 7.7).

4. Maintain lateral connectivity between riparian zones with adjacent habitats.

   There is a range of terrestrial and semi-aquatic species that require resources from terrestrial and riparian/aquatic components of the landscape in order to persist (Section 6.4).

5. Maintain longitudinal connectivity within river networks

[3] i.e. bankside

Many aquatic species either migrate or disperse up- or down-stream in order to complete their life cycles (e.g. some migratory fish species). Some amphibious, semi-aquatic or terrestrial species may also require longitudinally connected riparian corridors to persist in the landscape (Section 6.2).

6. Ensure that refuges and habitats for threatened species and narrow-range endemics are retained.

   No generalised scheme of managing a "mosaic" will capture all the biogeographic and evolutionary peculiarities of a region, specific amendments to standard prescriptions will be necessary to protect listed, threatened species and other species or groups of species deemed to be of conservation significance (Section 8).

3. Extended summary

3.1. A flexible, whole-of-catchment approach is needed to manage aquatic values

Many codes of practice for land management focus on mitigating impacts on aquatic systems via buffer or filter strips in the riparian zone, and modifying catchment activities to minimise inputs of pollutants (e.g. pesticides, sediments). Forestry research has also focussed on road-crossings and associated drainage systems as worthy of particular attention. While these measures ostensibly manage water quality, the conservation of aquatic biodiversity makes additional demands on land managers. Connectivity, both within the stream network and with the adjacent riparian zones, needs to be maintained or restored to ensure the persistence of migratory species in the landscape. There is increasing evidence that streams and their riparian zones subsidise food and resources to a range of terrestrial taxa, and that riparian zones may be more important to terrestrial systems than previously thought. Moreover, some parts of some catchments seem to be long-term refuges for narrow-range endemic taxa or provide refuges for threatened species, and catchments with extensive limestone (karst) areas provide additional challenges because of their subterranean stream systems.

Recent research suggests that catchments generate runoff patchily, and there are often "source areas" within a catchment that supply most of the overland flow to a system during wet periods. Managing activities on these source areas and similar catchment features may prove to be more effective at minimising inputs of sediment, nutrients and pollutants than simple "buffer" prescriptions (Croke and Hairsine, 2006).

Accordingly, a whole-of-catchment approach is needed to ensure the conservation of biodiversity in aquatic systems. At the landscape scale, tools such as CFEV in Tasmania provide a useful framework to facilitate this. There is likely to be much regional, biophysically-generated variation in the way aquatic ecosystems respond to forest management, and so the management framework for protecting aquatic biodiversity will need to be flexible to accommodate the biogeographic and physical peculiarities of parts of the Tasmanian landscape.

3.2. Issues for larger streams and waterbodies

The cumulative impacts of forestry activities in the upper parts of river catchments is difficult to evaluate, but two complementary projects are under way in Tasmania that could provide insights into the relative contributions of forest and non-forest land uses to downstream biodiversity and in-stream habitats (P.E. Davies, Freshwater Systems Pty Ltd and S.A. Munks, Forest Practices Authority Tasmania, pers. comm.).

Because fish are more prevalent in downstream reaches, barriers to upstream and downstream dispersal need to be minimised. Local research provides cost-effective solutions for fish passage through culverts. There may be cases where anthropogenic barriers have provided a refuge for native fauna (e.g. Swan galaxias) from introduced predators (e.g. brown trout, rainbow trout), in which case the barriers should be retained or other management actions undertaken as advised by the relevant agency or instrument (e.g. management plan, threatened species recovery plan)

The current streamside reserve prescriptions on Class 3 and larger streams are likely to provide shading and allochthonous resources to in-stream food webs, and there is no compelling evidence suggesting that these buffers need to be increased in width for these purposes, although burning can compromise the continuity of these reserves. Accordingly, more attention needs to be paid to quantifying the effects of escaped regeneration burns on in-stream biodiversity and habitats and the effectiveness and feasibility of changed burning practices by forest managers.

Streamside reserves are unlikely to provide complete protection from material transported by overland flow events (sediments, adsorbed nutrients and pollutants) or from materials delivered by poorly protected drainage lines that cross the buffer. Alternative strategies may better mitigate these effects at source; these include improved management of flow paths[4] created by forestry activities within the coupe and identification and protection of source areas of water and sediment.

Expansion of buffer zones may need to be considered on floodplain-like features that support water-dependent, non-instream taxa such as some frog species and burrowing crayfish.

Streamside reserves provide other ecosystem values beyond stream protection. There are semi-aquatic and terrestrial taxa that depend, in part, on secondary production from streams, and there are some taxa that are riparian specialists. Conversely, there are some taxa which appear to avoid riparian areas, and care needs to be taken to ensure that streamside reserves are not used as the sole means of maintaining terrestrial biodiversity in the landscape.

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[4] "Flow paths" refers to the small, numerous and often indistinct depressions and drainage lines that collect and concentrate overland flow during wet conditions. Flow paths can be both natural and anthropogenic in origin.

3.3. Issues specific to small, headwater streams

The biodiversity of small, headwater streams (loosely the Class 4 streams of the *Forest Practices Code* (Forest Practices Board, 2000)) is characterised by narrow-range, endemic taxa in some, but not all parts of Tasmania. Other small, headwater streams appear to have a fauna dominated by a subset of predominantly depositional taxa that are also present further downstream. However, the cryptic nature and taxonomic difficulties associated with this fauna argues for a precautionary approach. Better understanding of which catchments afford refuges for freshwater-dependent taxa during glacial maxima could provide the framework for more targeted comparisons of the potential for narrow-range endemism in Tasmania's river basins. This could lead to a "risk-based" set of criteria where headwaters in basins with demonstrated refugial characteristics are given a higher level of protection than in those that do not provide such refuges.

Small headwater streams are predominantly free of fish in Tasmania, but may still provide peripheral foraging habitat and burrowing opportunities for platypus. The importance of these streams for herpetofauna remains poorly investigated.

Functionally, small headwater streams are intimately connected with their adjacent hill-slopes, and forest harvesting increases surface run-off, channelizes the streams, and reduces standing stocks of organic matter. In the short term, in-stream photosynthesis is increased, and a number of metabolic changes occur consistent with infilling of interstitial habitats with fine sediments. Although in-stream photosynthesis can return to pre-harvest levels with 14 years in at least one forest system in Tasmania, many of the other metabolic changes persist. Longer-term recovery trajectories remain unknown.

The current Machinery Exclusion Zone (MEZ) of the Forest Practices Code applied to small, headwater streams probably affords some degree of protection for some components of in-stream biodiversity (e.g. hydrobiid snails that need hard surfaces supporting biofilm to feed on). The new, recommended Class 4 Guidelines (McIntosh and Laffan, 2005) provide a notable example of a flexible, risk-based approach that affords greater protection from erosion for headwaters in vulnerable catchments, with likely benefits to conserving biodiversity by virtue of better protection of in-stream habitats.

MEZs were not designed to "filter" sediments transported by overland flow events, and overland debris flows are relatively rare in Tasmania's forest estate. However, the MEZ is difficult to protect from burning where this is part of the management regime. Moreover, the (sometimes dead) trees remaining in the MEZ are more exposed to wind throw than when the forest canopy was continuous. The soil and bank disturbance resulting from increased wind throw could deleteriously affect in-stream habitat.

Experiments overseas investigating varying streamside reserve widths on small headwater streams are few, and in their infancy. Thus far the results are highly variable, and it should be acknowledged that no buffer width will completely protect a small headwater stream from the effects of intensive forestry operations. Forest managers and aquatic ecologists need to engage more openly with the issue of how much change is tolerable while maintaining biodiversity values in production forests.

Managers of forested headwater streams in Tasmania face some other challenges unrelated to harvesting and regeneration. These include the continued expansion of introduced lyrebirds (which can dig up large areas of headwater stream bed), likely increases in the frequency and intensity of wildfires, and the as yet unknown impacts of impending climate change on the frequency and intensity of rainfall events.

4. Introduction

4.1.1. Brief

This background document has been prepared to address the following:

To provide a Background document that reviews stream ecosystem management issues relevant for biodiversity conservation to assist the Biodiversity Expert Review Panel. The review will be limited to that part of item 2b of Terms of Reference (TOR) highlighted below in bold. In particular, the document should consider the research undertaken to address issues raised in the last review of the Code, relating to the management of stream biota, and translate outcomes into recommended actions.

Terms of Reference item 2:

2. Review the relevance and scope of the Forest Practices System in relation to biodiversity conservation and evaluate the ability of existing provisions to meet conservation objectives at the local, catchment and regional scales. In particular consider:

b. Processes and planning tools to address current forest practices at both the landscape and stand level. Provisions to address plantation design and planning are a priority. **Provisions for stream fauna are also a priority. In particular, consider the research undertaken to address issues raised in the last review of the Code, relating to the management of stream fauna, and translate outcomes into recommended actions.**

In order to address stream biodiversity adequately, issues surrounding ecosystem and habitat processes that sustain aquatic biodiversity have also been encompassed in this background document. This is necessary because forestry operations potentially affect the supply of sediment, nutrients and light to streams, and the flora and fauna of streams respond to these inputs. However, this review does not attempt to synthesise this information in terms of drinking water quality or similar human-centred valuations of water as an ecosystem "service"), nor the impact of forestry on the economic and social aspects of water yield, river erosion, flood frequency or water supply.

4.1.2. Background to research engendered by the previous review of the Forest Practices Code

The review of the 1993 Forest Practices Code (Forestry Commission of Tasmania., 1993) identified the issues concerning the management of Class 4 Streams as a priority for research, and accordingly research on small, headwater streams is emphasised in this review.

At the time of the previous review, there was much research focussed on the impacts of roading for forestry on streams, and this topic is also covered in some detail.

4.1.3. Arrangement of this document

My review of the literature on the effects of forestry on streams suggests that the biodiversity issues for small, headwater streams have some distinct issues that distinguish them from larger (usually permanent) watercourses and water bodies. Accordingly Section 5 sets the scene by describing the general effects of forest management on water quantity and quality as they pertain to biodiversity values. Section 6 describes the general, forestry-related biodiversity issues for streams and water bodies, while Section 7 describes issues specific to small headwater streams. Section 7 contains an admittedly ad hoc collection of other biodiversity issues that did not fit with this subdivision of categories.

4.1.4. Classification of surface waters by the Forest Practices Code

Table 8 on p. 56 of the *Forest Practices Code* (Forest Practices Board, 2000) is reproduced as Table 1 below, and these operational definitions will be followed in this review.

Table 1 Minimum streamside reserve widths or Machinery Exclusion Zones reproduced from Table 8 of the *Forest Practices Code*. Section numbers referred to in the footnotes refer to sections in the *Forest Practices Code* (Forest Practices Board, 2000)

Watercourse Type	Minimum horizontal width from watercourse bank to outer edge of reserve
Class 1. Rivers, lakes, artificial storages (other than farm dams) and tidal waters(1) generally those named on 1:100,000 topographical series maps.	40 m
Class 2. Creeks, streams and other watercourses from the point where their catchment exceeds 100 ha(2).	30 m
Class 3. Watercourses carrying running water most of the year between the points where their	20 m

catchment is from 50 to 100 ha[2].	
Class 4. All other watercourses carrying water for part or all of the year for most years[3].	Machinery exclusion zone: - no machinery within 10 m of streambanks except as below[4].

Notes:

1. Taken to be within 40 m of the high tide mark of tidal waters.

2. All catchment areas are to be confirmed on a 1:25,000 map prior to classifying watercourses. In upper catchments the Forest Practices Officer will assess the boundary between Class 1 and 2 watercourses based on local catchment conditions.

3. A Class 4 watercourse is differentiated from a drainage depression (see Glossary) by having at least one of the following features:
• a gravelly, pebbly, rocky or sandy bed, indicative of flowing water;
• an obvious gully;
• a short steep section of streambank adjacent to the watercourse bed.
A Class 4 watercourse will often have a change in understorey vegetation from the streambank to the surrounding forest e.g. riparian/moist vegetation on streambanks – ferns, mosses, sedges.

4. Harvesting machinery is permitted within 10 m under certain conditions at defined crossing points, and to undertake thinning (see Section C3.1). Conditions for the use of feller bunchers operating inside Class 4 machinery exclusion zones are detailed under Section C4. Site preparation machinery is permitted within 10 m under conditions detailed in Section E1.2.2.

Class 4 watercourses may be upgraded to Class 3 by a Forest Practices Officer depending

on local site conditions, particularly in eastern parts of Tasmania prone to high intensity rainfall.

5. Forestry, water quantity and quality

5.1. Scope

Forestry affects the delivery of water to aquatic systems and the chemical composition of the water. In this portion of the review, I focus on those aspects of water quantity and quality that have implications for biodiversity. The implications of forestry for water yield and water quality for human consumption and other human uses (e.g. stock water quality) are beyond the terms of reference for this review.

5.2. Water quantity

5.2.1. Qualitative changes in flow yield and pattern

For mature forests in long-rotation, native forest regeneration systems, harvesting generally results in an initial increase in water yield followed by a decline as the young trees regenerate and grow rapidly. As the stand grows older, water yields gradually return to pre-harvest conditions. This is depicted in Figure 1 for *Eucalyptus regnans* in central Victoria. Kuczera's curve (1987) is widely cited, but Watson et al.'s (1999) more comprehensive research documented an initial increase of water yield immediately after harvesting that was not documented in Kucera's early research. Evapotranspiration by the regenerating vegetation is the primary cause of decreases in streamflow. The leaf area index (LAI) of the canopy of the regenerating forest is a major contributor to the changes observed in empirical data, but other mechanisms (e.g. changes in conductance of leaves as trees mature) need to be invoked to match modelled predictions with empirical observations (Watson et al., 1999).

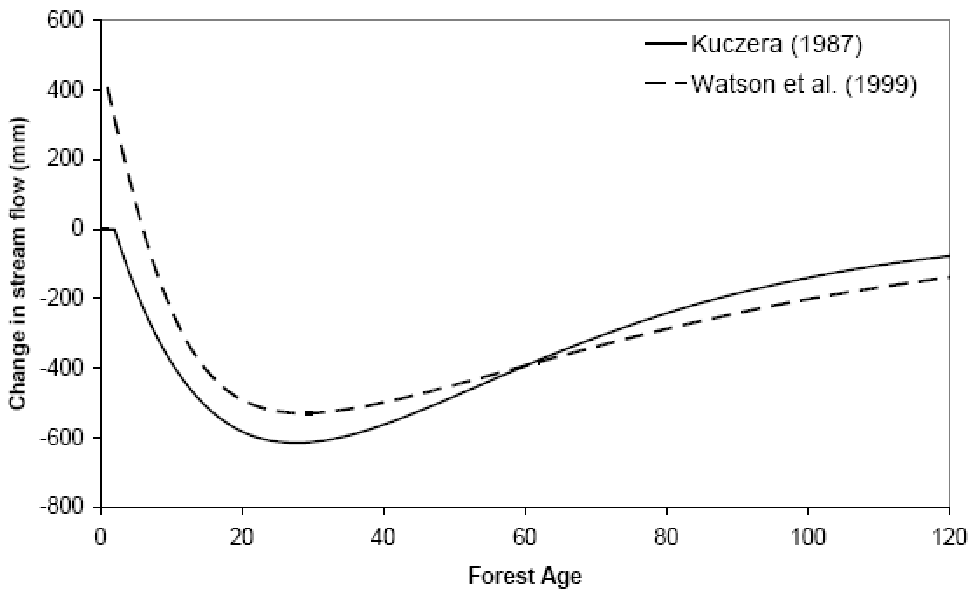

Figure 1 Change in streamflow (relative to pre-harvest conditions) as a function of forest age developed for *Eucalyptus regnans* forests receiving 2000 mm / y precipitation. (Figure taken from Brown et al. (2006).)

While the scenario in Figure 1 has received wide publicity, substantial deviations from it can be expected depending on the following factors (Vertessy, 1999, Schofield, 1996, Brown et al., 2006, Vertessy, 2000):

1. **Harvesting system.** Figure 1 applies to a clearfell-burn-and-sow (CBS) system. Less intense harvesting systems (e.g. shelterwood or selective cut) would result in less marked responses. Note also that similar responses can result from natural disturbances that kill much of the forest cover on a catchment (e.g. wildfire)

2. **Management of the forest on the catchment.** Any intervention that affects the LAI of the forest could affect the shape of the response in water yield. Different species of trees may have different physiological responses to rainfall and droughts, and these interspecific differences can further alter the shape and timing of the response curves depicted in Figure 1.

3. **Conversion of the catchment from one land use to another.** Conversions of forest to pasture or other non-forest agricultural uses of pasture usually results in large increases to water yields, whereas conversion of pasture to plantation often decreases yield.

4. **The nature of the catchment, including its relationship with groundwater.** Regional groundwater dynamics can either exaggerate or attenuate responses of a catchment to management of its vegetation cover. Some geological features (e.g. karst) can have profound effects on the way the surface waters in a catchment respond to precipitation.

5. **Stochastic events and climate change.** Bushfires are a prominent disturbance event in temperate Australian forests, and naturally occurring wildfires can be intense and result in major changes in streamflow. The climate of temperate Australia is notoriously variable, and decadal changes in precipitation inputs can affect yields directly via changes in runoff, with longer-term, lagged changes in supply from groundwater impacts being particularly marked in some regions of Australia (e.g. the south west of Western Australia). The implications of anthropogenic climate change present a major challenge to catchment modellers.

While changes to annual yield of water matter to those who manage water consumption, the timing and magnitude of flow events within a year are of more significance to biodiversity conservation. Figure 2 provides an example of the changes in the flow duration curve that have been observed during the early development of a pine plantation in a catchment near Tumut in New South Wales. Flow volume is represented on a logarithmic scale on the ordinate, while the percentage of time that a given flow is exceeded is represented on the abscissa. High flows occur on the left of the graph, low flows on the right. In this example, very low flows (< 0.01 mm/d) occurred *ca.* 10% of the time 1 year after planting, but were much more common (*ca.* 65%) 7 years later.

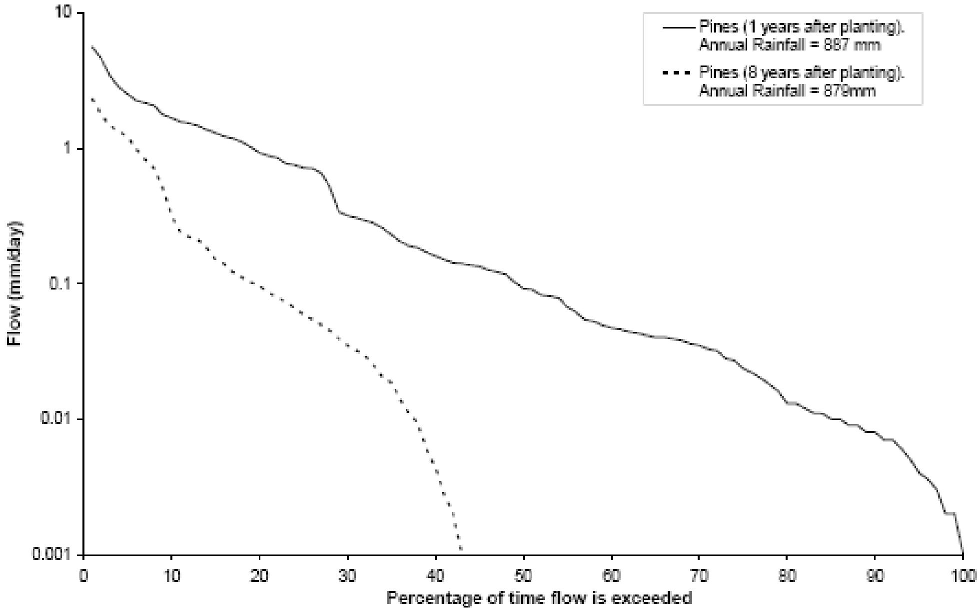

Figure 2 Flow duration curves for a catchment in NSW with 1 year old pines (solid line) and seven years later (broken line). (Figure from Brown et al. (2006).)

Changes in vegetation cover, including those resulting from changes in forest management, can have profound effects on the number and duration of low flow events and the duration and magnitude of high flow events. The caveats for the effects of forestry on water yield also apply to flow duration curves: less intensive management may attenuate responses, and the responses may be heavily modified by the local particularities of catchment lithology and geology and connection with groundwater.

Flow duration curves may also be too coarse to represent fully the potential impacts of forestry on biodiversity: the timing and sequencing of flow events is not represented, although these issues are most likely to be important in catchments where irrigation and other human manipulations of the direct flows in the river system alter the seasonal delivery of high flows to downstream reaches. In Tasmania, I surmise that these sorts of interventions are not made for the purposes of forest management, but note that other water users (e.g. irrigators, hydro electricity generators) can have profound effects on the timing and sequencing of peak and low flow events in some Tasmanian catchments.

5.2.2. Implications for biodiversity

Clearly the effects of forestry depend on a number of important contextual issues in any given catchment, and blanket pronouncements even on qualitative aspects of forest management will likely prove too vague to be useful in managing biodiversity. Following Brown et al. (2005), Brown et al. (2006) and Vertessy (2000), it is more tractable to consider changes to flow patterns from the perspective of the major types of forest management that prevail in Tasmania.

Table 2 attempts to summarize the qualitative evidence for changes in flow type and pattern for the three broad scenarios of forest management in Tasmania considered for the TasLUCaS[5] models of Brown et al. (2006). Empirical evidence for changes due to conversion to native eucalypt plantation species is currently very sparse, and Vertessy (2000) emphasised the variable and often statistically weak data concerning changes to flow patterns (as measured by flow duration curves) that are relevant to temperate Australian conditions.

Direct empirical data on the responses of in-stream biota to water yield and flow duration as wrought by forestry management are very scarce. The majority of the literature focuses on the more proximal effects on in-stream and riparian habitat shortly after harvest (e.g. Growns and Davis, 1991, Davies and Nelson, 1994), and much of the in-stream research to date has concentrated on evaluating riparian prescriptions to ameliorate the short-term, local impacts of forestry on the biota (e.g.Davies and Nelson, 1994).

There are no long-term studies that strongly link changes in the in-stream biota to changes in flow regime from forest management *per se*, and it would be difficult to envisage how such studies could disentangle forestry management from confounding local and regional changes (e.g. climate variability, stochastic events such as intense floods, droughts, wildfires). Similarly, it is hard to separate forestry from other land use changes when trying to judge cumulative changes that manifest themselves in the downstream reaches of catchments, and there are no published studies comparable with Tasmanian conditions that provide any convincing evidence that unconfounds these changes.

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[5]Tasmanian Land Use Change and Streamflow

Accordingly, the cumulative impacts on downstream reaches that are sketched in Table 2 are speculations based on the outputs of modelling exercises such as TasLUCaS. For example, water abstractions during summer for irrigation can lead to much lower base flows and reduce fish passage opportunities for summer-dispersing species. By analogy, a catchment with a large proportion of its area subject to intensive management for short-rotation plantation forestry could reduce summer flows in downstream reaches and pose similar problems as other abstractive uses of water. Whether the magnitudes of these effects for plantation forestry approach those of irrigation uses remains moot, thereby emphasising the usefulness of spatially explicit modelling tools such as TasLUCaS and WAFL[6] to evaluate the relative contributions of different land uses and temporal changes in the mix of these land uses for flow timing and volume. A retrospective evaluation of cumulative impacts of progressive harvesting of pine plantations in New Zealand found some patterns of in-stream biodiversity related to the harvesting regime, but other differences in the catchment types and land uses dominated the differences in community composition between the catchments (Collier and Smith, 2005).

Conversion of cleared agricultural land back to some form of forest cover may provide some benefits to biodiversity (Brockerhoff et al., 2008). The initial clearance of forest to provide pasture or cropping usually leads to marked changes in flow regimes: previously temporary or ephemeral streams become permanent, and flows often become "flashier" (i.e. peak flows are greater, but of shorter duration). The intensification of high flows usually increases the competence of the stream to move bed materials, which, in turn, leads to increased erosion and incision of the stream bed. These changes are sometimes dramatic and irreversible (Brierley and Fryirs, 2005, McIntosh, 2007, McIntosh et al., 2007). Restoration of forest cover in such catchments can ameliorate these changes to flow regime, but some of the very large habitat changes that happened while the catchment was under agricultural use may not be reversed without direct intervention by humans with in-stream works (Brierley and Fryirs, 2005).

From a Eurocentric perspective, reforesting previously cleared catchments, therefore, can have the apparently perverse outcome of increasing the ephemerality of streams in a catchment. It is important to realise, however,

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[6] Water Availability and Forest Landuse Planning Tool, currently being developed by DPIW for regional planning purposes. It is based on CFEV and some components of TasLUCaS.

that the “natural”, forested condition of some these catchments probably meant that the smaller streams in the network were ephemeral or seasonal in their flows, and that there is a flora and fauna that is adapted to such conditions, often relying on cease-to-flow events to persist in the landscape. The notion of restoring ephemerality to streams that have been made permanent by land clearance remains little investigated for forest systems.

Nevertheless, not all small streams would have been ephemeral when forested, and the interplay of groundwater and surface water is complex. Clearly some species only persist in permanent freshwaters and are only found in headwater reaches of some Tasmanian streams (e.g. hydrobiid snails in the genera *Beddomeia* and *Phrantela*).

Table 2 Qualitative changes to flow yield and pattern under different broad categories of forest management likely to prevail in Tasmania. Changes are noted relative to the land use or forest cover before the management for forestry.

Type of forest management	Changes to water yield relative to pre-forestry land use	Changes to flow patterns	Potential problems for aquatic biodiversity	Potential benefits for aquatic biodiversity
Native forest harvesting and regeneration	In CBS systems, changes to yield follow form of Watson et al. (1999) in Figure 1. Timing and magnitude of the effects depend on the species and prevailing climate. Patterns could be much attenuated in shelterwood and selective cut systems.	Depends on age of regenerating forest. Initial intensification of high flow events immediately after harvest, shifting towards drier flow duration scenarios during most intensive phase of regeneration. Longer-term responses depend on forest management. Less intensive management systems may result in attenuated changes in	Habitat changes from initial increases in flow after harvest: inputs of fine sediment, increased competence of stream may increase channelization and change proportional representation of in-stream habitat types (Davies et al., 2005a, Davies et al., 2005c). Potential for increased frequency and duration of low flows and cessation of flow during years of intense water	Some indirect benefits (e.g. off-stream fire dams may provide drought refuge for amphibians).

Type of forest management	Changes to water yield relative to pre-forestry land use	Changes to flow patterns	Potential problems for aquatic biodiversity	Potential benefits for aquatic biodiversity
		flow pattern.	use by regenerating forest.	
Conversion of native forest to plantation	Depends on species involved. Pine plantations probably loose more water to evapotranspiration than native eucalypts, but there are insufficient data for eucalypt plantation species in Australia (Vertessy, 2000).	Depends on species involved. Pine plantations may shift flow duration curve to drier pattern than native eucalypt forest. Little empirical data available applicable to Tasmania.	In-stream habitat changes from increased runoff during establishment phase. Loss of habitat if plantation species or management increases frequency or intensity of low flow events during dry season.	
Conversion of pasture or cropped agricultural land to plantation	Decrease in yield relative to agricultural conditions	Depends on age of plantation and rotation times. General shift to lower flows that intensifies as water requirements of trees increase. Low flows	Dewatering of smaller streams during dry spells and seasons. Cumulative effect downstream could compromise longitudinal	Return to a pattern of flows more similar to that prior to agricultural use. Restoration of native riparian species to

Type of forest management	Changes to water yield relative to pre-forestry land use	Changes to flow patterns	Potential problems for aquatic biodiversity	Potential benefits for aquatic biodiversity
		tend to be more frequent and more prolonged; high flows less intense (Vertessy, 2000). Longer term-responses poorly documented with empirical data relevant to Tasmania.	connectivity (e.g. fish passage) in river segments remote from forestry activity.	stream sides.

5.2.3. Conclusions

The largest gap in our knowledge about the impacts of hydrographic changes from forestry relate to the cumulative impacts of forestry on the downstream reaches which are usually remote from the forestry operations themselves. It will remain difficult to disentangle the impacts of forestry from those of other land use changes in the catchment, and datasets are usually too short to encompass large, extreme events (Kaimowitz, 2004). Accordingly, modelling tools such as TasLUCaS and WAFL hold the most promise for assisting water managers to compare the effects of different mixes of land-use changes on flow yield and pattern. Combining such tools with CFEV should help prioritize those catchments for which flow changes attributable to forestry are most likely to affect biodiversity deleteriously.

That said, modelling tools are only as useful as the data on which they are based, and the deficiencies in the current tools need to be examined critically, and empirical studies should be targeted at the major information gaps. Changes to yield and flow duration patterns resulting from conversion to native plantation species appears to be a prominent gap.

In the medium to longer term, it will probably be beneficial to base forest management on more proximal measures of water use (e.g. LAI) rather than the coarse approximations implied by using stem densities or proportion of catchment subject to a given forestry regime. This, of course, relies on the development of cost-effective means of estimating such proximal measures (e.g. by remote sensing).

The biota of ephemeral and temporary streams in Tasmania's forested landscapes remains poorly documented, and this remains a major gap in our knowledge of biodiversity values, and the potential biodiversity benefits that might result from reforesting previously cleared land.

5.3. Water quality

Much of the research on forestry impacts on water quality focus on aspects of interest to consumptive uses of water for humans (e.g. clarity and turbidity, salinity), which has resulted in a strong bias towards solutes dissolved or in suspension in the water column. Most of the in-stream biota of rivers and streams is benthic (i.e. bottom-dwelling), and it is the material sorbed to particulates and likely to deposited in the sediments that is most likely to affect aquatic biodiversity (Campbell and Doeg, 1989), although Growns and Davis (1991) attributed some of the differences they observed in clearfelled catchments in Western Australia to increased salinity which they presumed resulted from forestry. There are two major groups of chemicals from forest management that cause concern from a biodiversity perspective: nutrients and pesticides. Large components of both arrive in the stream sorbed to sediments, and increased sediment inputs have physical impacts on in-stream biota by themselves (Campbell and Doeg, 1989, Gillespie, 2002, Fortino et al., 2004, Trayler and Davis, 1998).

5.3.1. Impacts of sediment

The effects of sediment from forestry operations are well documented. They include infilling of interstices by fine sediment, which can alter benthic diversity community composition (Campbell and Doeg, 1989, Growns and Davis, 1991, Growns and Davis, 1994). There are even effects on the rarely examined meiofauna in sandy-bottomed streams, although Trayler and Davis (1998) were unable to attribute the logging-induced changes exclusively to infilling by fines from timber harvest in their study in Western Australia. The recruitment and growth of some fish and amphibians are deleteriously affected by sediment inputs (Campbell and Doeg, 1989, Gillespie, 2002, Hemstad and Newman, 2006) by a variety of mechanisms: smothering of eggs, reduction of appropriate habitat, increased oxygen demand from organics transported into the stream after harvest and more subtle effects resulting from the changes in the configuration of the food web (England and Rosemond, 2004).

There are also probably cumulative downstream impacts (e.g.McIntosh, 2007, McIntosh et al., 2007), especially on receiving water bodies such as wetlands and estuaries. Separating the effects on biodiversity of forestry from those of other contributors such as agriculture and urbanization remains problematic, however.

Croke and Hairsine (2006) argue that better management of sediment inputs could be achieved by more careful consideration of “source areas” and delivery pathways (or “flow paths”) in a catchment and affording those more protection.

“Source areas”, loosely, refer to those (terrestrial) parts of the catchment that tend to yield most of the water that flows into a stream. Catchments are not uniform in their behaviour as they get wetter during a prolonged period of precipitation, and so some parts of the catchment are more likely than others to generate overland flow. “Flow paths” refer to the numerous, often indistinct, depressions and minor drainage lines that can fill with water and connect with each other after heavy rain. Longer, low-gradient the flow paths are more likley to deposit sediment before it reaches the stream network proper. A combination of protecting these source areas and managing the flow paths leading from them is likely to be a more effective means of reducing forestry-related impacts than solely relying on streamside reserves and elegant road crossings. While much progress has been made in managing anthropogenic flow paths over the last 30 years (e.g. snig-tracks, log-landings, wheel ruts, routing of drains), the challenge will be to identify the more subtle, often natural features prior to harvest and develop realistic procedures to manage them. As ever, empirical tests of the efficacy of these measures on biodiversity have not been attempted.

Substantial progress has been made in reducing sediment movement off of timber harvesting coupes. Strategies include various streamside reserve or buffer prescriptions, better management of road drains and crossings and improved harvesting practices and management of drainage lines (e.g. Hairsine et al., 2002). The objective is to use a combination of measures to increase the length of the flow paths across forestry coupes so as to maximise the retention of soil and nutrients in the coupe (Croke and Hairsine, 2006).

5.3.2. Nutrients

Nitrogen and phosphorus are the major nutrients likely to be mobilised during forestry operations, with the bulk of the inputs occurring within the first few years after harvest (Campbell and Doeg, 1989), although responses to forestry can be highly variable, and are often confounded with catchment differences and stochastic events such as wildfire and high flow events (Olive and Rieger, 1987). Occasional additional inputs of nutrients may also result from fertilization in more intensive silvicultural systems.

Transport of nutrients into streams is usually episodic, with much of the load being delivered by a small number of large rainfall events that generate the biggest overland flow events. This is because most of the nutrients are sorbed to particles (Woodfull et al., 1993). Accordingly, managing the movement of sediment off the land surface of a coupe is usually regarded as the most important step in minimising export of nutrients from the coupe (Campbell and Doeg, 1989, Croke and Hairsine, 2006).

The transformation of nutrients as they move across the catchment and through the interstices of the soil can be complex. Recent fine-scale research in riparian zones of boreal Canadian streams suggested that saturated patches of soil can denitrify water before it reaches a stream (Luke et al., 2007). Furth more, because catchments are spatially heterogeneous in terms of their yield of both water and sediment, Croke and Hairsine (2006) call for a shift in management away from simple buffer strip prescriptions to explicitly managing flow paths to retain nutrients and sediments on the catchment land surface.

5.3.3. Pesticides

Pesticides have variable impacts on in-stream biota. Inputs of insecticides, predictably, have more marked impacts on in-stream invertebrates than herbicides (e.g. Davies and Cook, 1993, Davies et al., 1994), and practices with the use of pesticides in forest management have improved greatly over the past three decades (Reichenberger et al., 2007).

More recently, concern has focussed on pesticides and their residues that are capable of disrupting the endocrine systems of animals ("endocrine disrupting chemicals" or EDCs), and some of these pesticides are commonly used in plantation establishment in Tasmania. Overseas, the effects on aquatic vertebrates (e.g. fish, amphibians and reptiles) have been more intensively researched than in Australia (Fox, 2001, Williams et al., 2007, Campbell et al., 2006). There is some very recent European research showing effects of EDCs on introduced hydrobiid snails in sediments contaminated by coal mining effluents (Mazurova et al., 2008), so there is some potential for these chemicals to similarly affect closely related Tasmanian endemic species. Very little of this literature addresses EDCs from forestry-derived sources, and most of the well-established cases concern point-source effluents from sewage treatment plants, industrial facilities, dairies and other food-processing plants, and the intensification of urbanisation. The recent Australian study by Williams et al. (2007), did not address forestry explicitly, although diffuse rural sources of EDCs in agricultural areas were identified as a concern. Similarly Fox's (2001) Canadian review mostly cites examples from industrial and agricultural locales, but he cautions against complacency given the patchy attention given to monitoring EDCs (see also Campbell et al., 2006). Reliable methods to screen for EDCs remain to be developed, although Campbell et al. (2006) outline several possibilities. The application of pesticides in forest management will likely lead to much lower concentrations of EDCs than documented in these reviews; however, all reviewers not the potential for these compounds to accumulate in downstream sediments, and their strong propensity to bioacculate and magnify up food chains. Consequently, a "watching brief" on this emerging area would be wise, and the risks posed by EDCs from forestry-related pesticides and chemicals as they are currently used needs to be assessed.

Apart from avoiding direct application of pesticides into streams (e.g. avoiding aerial spraying over stream lines), buffer strips have been one of the main management tools used to retard or prevent pesticides moving into waterways. The results have been mixed, and climatic factors (e.g. rainstorms) and minor drainage lines can lead to breaches of riparian buffers, with sometimes catastrophic results (Reichenberger et al., 2007, Lacas et al., 2005) (Norris, 1993).

Pesticides vary in their toxicity and mobility, and these two axes of variation have been used to develop a Pesticide Impact Rating Index (PIRI)[7] which provides a "first tier" risk assessment tool to support decisions about pesticide use in a given circumstance. Thus a pesticide might be judged to be moderately toxic to aquatic life, but it may sorb strongly to soil particles in the catchment being treated and it might be unlikely that these particles would be transported into surface waters before the pesticide degraded. A version of PIRI is being developed for Tasmanian conditions, and field and laboratory research has added important additional information about the behaviour of pesticides in Tasmanian soils (R. Doyle, Tasmanian Institute of Agricultural Research, pers. comm.).

The development of such risk-based decision tools opens the possibility of tailoring pesticide prescriptions to the specifics of a catchment and the prevailing climate. Further validation, including field-based testing of PIRI in forest management is warranted. However, the advent of such tools promises a much more focussed, and locally appropriate method of modifying prescriptions in the future.

5.3.4. Conclusions

Focussing on the concentrations of nutrients and pesticides in the water column of streams is unlikely to yield information of much value for biodiversity conservation by themselves. Sediment inputs are the more proximal mechanism by which catchment disturbances will affect aquatic biodiversity values.

7 http://www.clw.csiro.au/research/biogeochemistry/assessment/projects/piri.html

6. Forestry and aquatic biodiversity

6.1. Scope

This section includes all rivers, larger streams (Class 3 and larger under the Forest Practices Code), wetlands and anthropogenic water bodies such as fire dams because, until recently, much of the research into forestry impacts ignored small headwater components of the drainage network, and there remains little research into temporary aquatic systems in forested areas (Campbell and Doeg, 1989)

6.2. Roading and barriers

Roads and their associated drains have received much attention because they may be substantial sources of sediment input into streams, although this is not always the case (Sheridan and Noske, 2007 and references therein). The impacts of roading are often linked with those of other tracks and track-like disturbances in forest harvesting, and these issues are treated in more depth for headwater streams in Section 7.4.1.

Culverts and similar structures used for road crossings over streams present the biggest hazard to dispersal of fish. Macdonald and Davies (2007) have demonstrated that a modest increase in roughness in culvert pipes achieved using inexpensive spoiler baffles improved the passage of two species of diadromous native galaxiid fish by up to 80%. This simple measure would improve longitudinal connectivity in streams in Tasmania, provided a comprehensive field manual was developed to integrate this with other design and operational requirements for stream crossings. For example, many smaller streams are naturally fish-free and too steep to allow fish to move upstream, so engineering for fish passage is unlikely to be useful.

Other barriers to fish dispersal include weirs and similar structures and there is a wide variety of fish passage solutions available in the engineering literature, although there is a dearth of quantitative information about swimming speeds and dispersal requirements of native Australian fish (Macdonald and Davies, 2007 and references therein).

The only caveat to routinely ensuring longitudinal connectivity for fish dispersal is in situations where anthropogenic in-stream barriers have prevented dispersal by exotic fish species, thereby creating a refuge for native species. Swan galaxias might be one example, since it is clearly unable to co-exist with introduced trout. It persists upstream of natural barriers to trout and is rapidly lost from reaches into which trout are (illegally) introduced. Since the full extent of its range has yet to be established, there could be some streams that have inadvertently become isolated from trout invasion by the construction of in-stream barriers related to forest management.

For larger species, such as native fish, alterations to the general principle of ensuring fish passage can be captured on a case-by-case basis through such mechanisms as the Fauna Value Database maintained by the Forest Practices Authority. The extent to which naturally fish-free streams are preserved as free of fish by virtue of anthropogenic in-stream structures remains unknown and would form a valuable addition to the CFEV database.

6.3. Riparian zones

Most of the energy base for the aquatic food webs in forested catchments is derived from the riparian zone (Naiman and Décamps, 1997). As well as providing the main energy inputs, riparian trees and shrubs regulate light inputs and stream temperature. There is a copious literature documenting the importance of retaining riparian shade to keep stream temperatures cool enough for salmonid spawning, particularly in the Pacific Northwest of the U.S.A and Canada (Moore et al., 2005 and references therein). Woody vegetation in riparian zones is also crucial in providing large woody debris (LWD), which, in-stream, provides shelter and spawning habitat for some species of fish and crayfish as well as supporting often distinctive invertebrate and microbial communities. LWD also retains coarse organic matter and retards the downstream transport of both it and some nutrients so that more material is processed close to the site of input (reviews in: Lovett and Price, 2007).

The current prescriptions of the Forest Practices Code (Forest Practices Board, 2000) provide substantial protection to riparian zones for Class 3 and larger streams, and Tasmanian research based on space-for-time surveys of paired sites suggests that statistically defensible changes in macroinvertebrate community structure and algal biomass only occur when riparian buffers are < 30 m wide (Davies and Nelson, 1994). Curiously, this research found no effect of soil erodibility nor coupe slope on the biotic variables measured, which is at odds with the documented relationships between these variables and the inputs of fine sediment both in Tasmania (Davies and Nelson, 1993, Wilson, 1999) and elsewhere (Kreutzweiser and Capell, 2001). This study also found no recovery in affected streams within the maximum 5-year time period since harvest encompassed by the study, and similar slow recoveries have been suggested by Tasmanian research in granitic catchments of the north-east (Davies et al., 2005a, Davies et al., 2005c). Many other studies in temperate forest catchments substantiate the arguments for buffers or streamside reserves of similar widths to those prescribed by the Forest Practices Code (e.g. Boothroyd et al., 2004, Naiman and Décamps, 1997, Quinn et al., 2004, Moldenke and Ver Linden, 2007), so the cumulative evidence for retaining native riparian vegetation is strong.

Moreover there is a growing body of research reporting biodiversity benefits of re-establishing native vegetation in the riparian zones of cleared catchments (e.g. Teels et al., 2006) despite the young age of many riparian rehabilitation efforts. This argues for the re-establishment of local native species in the plantation estate where native riparian shrubs or trees were cleared under older management regimes. This coincides with recent conceptual models of restoring a matrix of patch types and corridors in managed landscapes (Munks and McArthur, 2000, Fischer and Lindenmayer, 2007, Fischer et al., 2006), although the spatial configuration of stream and cave networks provide particular challenges for future research in conservation planning (Campbell Grant et al., 2007, Fagan, 2002, Muneepeerakul et al., 2007) and conservation genetics (Hughes, 2007).

6.4. Catchment disturbance

Burcher et al. (2007) provided a schematic framework for conceptualising how different catchment elements and disturbances to them can "cascade" down to in-stream biota (Figure 3). They compared 10 similar-sized catchments on similar geologies in western North Carolina with the catchments differing in their mix of land covers (forest, agriculture and urban development). Using land cover as a proxy for catchment disturbance, Burcher et al. (2007) demonstrated that different elements of the cascade in Figure 3 were important to different components in the "response" communities. Interestingly forest cover was only important in one of the nine significant cascade models: urbanisation and some classes of agriculture had particularly marked effects that circumvented riparian elements in the landscape.

The particularities of these models are of less interest than the general principle: different land uses can affect in-stream biodiversity via different mixtures of pathways, and understanding the relative importance of the mechanisms in a given catchment context can shift the emphasis, say, from riparian management to managing erosional source areas on convergent slopes or diffuse pathways (Lane et al., 2004, Lane et al., 2006).

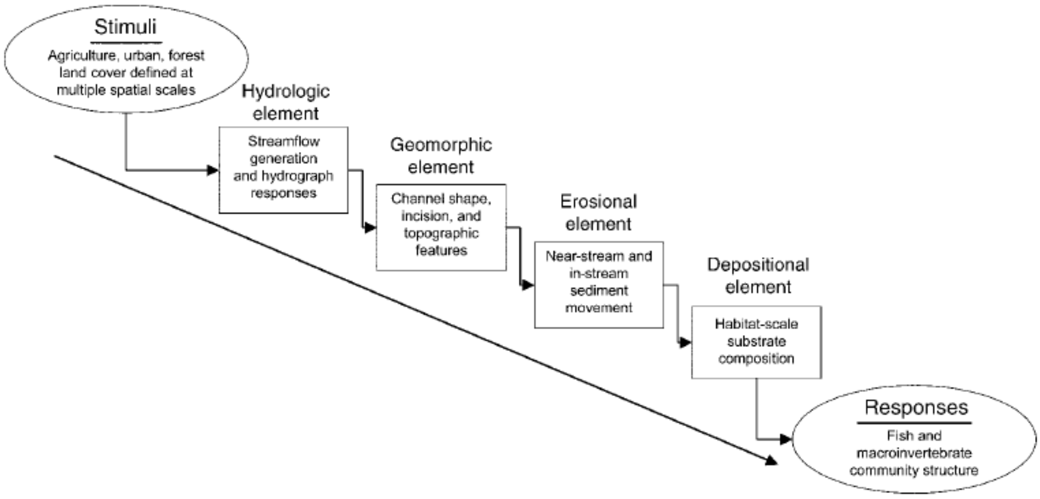

Figure 3 Schematic model of connections between catchment disturbances and in-stream responses to them. (Figure taken from Burcher et al. (2007))

In a sense, Burcher et al.'s (2007) framework is a biocentric generalisation of the conceptual framework proposed by Croke and Hairsine for managing sediment inputs to streams. They share important common features: the use of spatially-explicit, GIS tools to identify "risks" in any given catchment, and an acknowledgment that different catchments may need different management strategies to protect aquatic values.

However, forest practices are not as widely divergent as the land uses encompassed by Burcher et al. (2007), and a broad consensus of the literature suggests that maintenance of native riparian vegetation generally attenuates the deleterious effect of forest operations on in-stream biota and that disturbing this zone often results in detectable effects on in-stream community structure (Tomer et al., 2003, Anderson et al., 2007, Davies and Nelson, 1994, Kreutzweiser et al., 2005, Lovett and Price, 2007, Luke et al., 2007, Melody and Richardson, 2007). Riparian zones can get breached during intense rainfall events, and the changes to the catchment's flow paths from some harvesting operations can result in notable biotic and abiotic effects in spite of riparian protection (Croke and Hairsine, 2006, Campbell and Doeg, 1989 and references therein).

In some areas, catchments contain a variety of other aquatic and semi-aquatic habitats such as floodplain pools and wetlands, saturated zones, seeps and small temporary water bodies that are used by a variety of aquatic and semi-aquatic species. In a North American study, for example, a 30.5 m riparian buffer was insufficient to accommodate all of the requirements of the Boreal Toad (*Bufo boreas*). The use of seeps, drainage depressions and other semi-aquatic features outside the buffer varied seasonally, from year to year and even between the sexes. Some conservation of such habitats that fall outside buffers may be necessary, and recent proposals by the Forest Practices Authority (P. McIntosh, pers. comm.) for guidelines to protect drainage lines and depressions are likely to assist in this regard.

"Outside buffer" requirements are likely for a number of species. Frogs are a prominent example, with some species using edges or ecotones for dispersal (including those created by forestry Baker and Lauck, 2006), while others seek refuge in seeps and off-stream wetlands to complete their life cycles (Cushman, 2006, Olson and Rugger, 2007, Semlitsch and Bodie, 2003). Similarly, platypus are generally more abundant in larger streams and waterbodies, but some segments of the population may use small headwater streams at some times of the year (Koch et al., 2006). While they depend on aquatic invertebrates for their food, their burrows may be up to 200 m away from a stream bank (S. Munks, Forest Practices Authority, Tasmania, pers. Comm.).

Litter beetles provide another example: some are riparian specialists and may even have aquatic juvenile stages (e.g. the Psephenidae) and, as one moves away from streams and into the forest interior, there are overlapping suites of species, with some apparently actively avoiding wet habitats (Baker et al., 2006). Edges or ecotones in the forest can affect beetle diversity (Baker et al., 2007b), and riparian distinctiveness disappeared in headwater catchments in one Tasmanian study in tall wet eucalypt forests (Baker et al., 2007a).

Prior to harvesting, forested catchments in Tasmania were complex mosaics of habitats. While riparian zones are clearly crucial to in-stream ecosystem processes and biodiversity, there is an array of amphibious and semi-aquatic species that use both aquatic and terrestrial components of catchments to survive. These "lateral" connections can extend to surprisingly extensive subsidies of energy, not just from the terrestrial environment to the stream , but also from aquatic habitats to the terrestrial environment (Bastow et al., 2002, Marczak et al., 2007, Sabo and Power, 2002, Wipfli, 2005). Unfortunately, such studies have not been attempted in Australia, much less Tasmania, so quantifying the strengths of these lateral connections and their importancy for biodiversity is impossible. Instead, Fischer et al.'s (2006) "guiding principles" for conserving ecosystem function and biodiversity in production landscapes provide the qualitative framework for managing forestry-related catchment disturbances:

> *Landscapes should include structurally characteristic patches of native vegetation, corridors and stepping stones between them, a structurally complex matrix, and buffers around sensitive areas. Management should maintain a diversity of species within and*

across functional groups. Highly focused management actions may be required to maintain keystone species and threatened species, and to control invasive species.

7. Small headwater streams

7.1. Definitions

Class 4 streams form part of a "headwater stream network" which can be defined as:

> *The headwater stream network consists of first order, and where necessary, second order stream segments in a drainage network. It includes their sources or zero order basins where there are identifiable aquatic habitats for at least part of the hydrological cycle. Such headwater streams are closely coupled with hillslope and other adjacent terrestrial processes.*

The rationale for this definition is as follows. The source or zero-order basin of a stream is defined as "an unchannelized hollow with convergent contour lines" (Tsukamoto et al., 1982). Mostly such basins support terrestrial rather than aquatic life, but in some systems the zero-order basin gives rise to seasonal or ephemeral channels before coalescing into a landform that corresponds to any vernacular definition of a small, headwater stream. Most authors would regard these channels as being the first-order stream (Gomi et al., 2002), and it includes the upstream heterogeneous zone (UHZ) of Gooderham et al. (2007).

Many first-order streams are small and lack the hydraulic power to move the structural, bed-forming components of their beds and banks even at high flows (Gooderham *et al.*, 2007); some, however, are steep and carry substantial bedloads after intense rainfall events (e.g. McIntosh et al., 2007). All headwaters are strongly coupled with hillslope processes in areas of high relief, at least, and may have either perennial or sustained intermittent flow. Gomi et al. (2002 p. 908) define the latter as "more than 4 to 5 months during an average year", and since these researchers are working in the Pacific coastal forests of Canada and the U.S.A., this definition is broadly applicable to Tasmania owing to the similarity in rainfall patterns.

The inclusion of second order streams, where appropriate, is justified by Gomi et al. (2002), who note that second order streams are also closely coupled with hillslope processes in some systems, so that many of the processes that are important to first-order streams are also important to these segments of the network as well.

The current *Forest Practices Code* (Forest Practices Board, 2000) sets an upper limit on catchment size of 50 ha for Class 4 streams, with caveats for "upgrading" a stream's status to Class 3; these have been elaborated and extended by new guidelines (Forest Practices Board, 2004).

7.2. Distinctive features

7.2.1. Hydraulic features

Small headwater streams can comprise > 30% of the total length of the stream network in a drainage basin (Alexander et al., 2007). Even these figures are likely to underestimate the extent of first-order streams because many are unmapped (Alexander et al., 2007). In Tasmania, the proportion of drainage networks that could be described as "small headwaters" is likely to be even higher given the proximity of the heads of most catchments to the sea.

Despite their abundance, these streams remain poorly known biologically, and this prevails internationally as well as in Tasmania. However, recent research in Tasmania has documented some of the features of this part of the stream network in the forested areas (Davies et al., 2005a, Davies et al., 2005c, Koch et al., 2006, Clapcott, 2007), and recent research in forested catchments of western North America has led to a synthesis of features summarised in Table 3.

For forested headwaters in upland areas, the streams tend to be steep, with a stair-step longitudinal profile and are subject to unpredictable land-slips or debris flows. Hydrologically, the permanent streams tend to derive a greater proportion of their modal flows from groundwater than downstream segments, and water depths are shallow and velocities small (Gomi *et al.*, 2002). Because of their small size and large contact with the adjacent terrestrial habitat, flows are responsive to runoff events, resulting in flashy hydrographs, although the peak discharges obtained are rarely severe enough to move sediments much coarser than gravels in low gradient streams(Gooderham *et al.*, 2007), but there may be very large erosion events in steeper streams in areas prone to very intense rainfall events (McIntosh et al., 2007). Streams with groundwater input often have intermittent flows, and in drier climates, watercourses become ephemeral (Richardson and Danehy, 2007, Winter, 2007).

Table 3 Features of small headwater streams that distinguish them from larger streams and rivers. Modified from Richardson and Danehy (2007).

Feature	Headwater characteristics	Biological responses	References
Low hydraulic power	Most flows incompetent to move bed-forming materials.	Benthic habitats may be more stable. Highly retentive of CPOM. Greater proportion of habitats and biota (e.g. mosses) intolerant of scour.	(Gooderham *et al.*, 2007) (Watson, 2005) (Hassan *et al.*, 2005)
Geomorphology	Typically steep gradients, step-pool morphology, high stability	Stable habitat Small habitat volume	(Hassan *et al.*, 2005)
Flow paths and temperature	Groundwater inputs more important than advective inputs	Moderation of temperature, less variable process rates	(Gomi et al., 2006, Moore and Wondzell, 2005) (Moore et al., 2005)
Flow seasonality	Winter high flows, summer low flows; may dry seasonally	Prolonged high flows may inundate adjacent riparian zone which may augment metabolic inputs to stream food web. Drying of stream may have long-lasting effects on biota	(Clapcott, 2007) (Moore and Wondzell, 2005)
Disturbance regimes	Vulnerable to low flows, landslides or debris flows	Habitat loss and isolation	(Swanson *et al.*, 1998)
Recolonization pathways	From downstream; possibly between catchments for flighted forms.	Long path length; possible barriers	(Fagan, 2002)

Aquatic-terrestrial linkages	Highest edge:area ratio	Tightly integrated	(Nakano and Murakami, 2001)
Aquatic vertebrates	Typically fish-free because too shallow or flows too intermittent or because of natural barriers downstream. Platypus do access headwaters	Food webs have invertebrates as top-predator, although fish-free streams provide predator-free space for herpetofauna to fill role of top predator in North American systems.	(Richardson *et al.*, 2005) (Koch *et al.*, 2006)
Canopy closure	Restricts light reaching stream bed. Allochthonous inputs of organic matter, wood as retention structures.	Food webs based on detritivory and use of microbial biofilms. Highly retentive systems, with high storage of OM.	(Kiffney et al., 2003) (Watson, 2005) (Davies *et al.*, 2005a)

All these factors result in streams with low competence to move bed materials, and bed forms that result in highly heterogeneous habitats, with larger elements being stable for long enough to support the growth of mosses and other taxa which are usually scarce in larger streams with more extensive high flow velocities (Gooderham *et al.*, 2007).

7.2.2. Close riparian connections

The riparian vegetation usually forms a closed canopy, and most of the energy for the in-stream food web is provided by allochthonously-derived inputs of leaf litter (often termed CPOM: coarse particulate organic matter), while inputs of tree limbs and trunks (LWD: large woody debris) adds structure to the stream bed, which retains CPOM for processing close to the site of input (Gorecki et al., 2006). There are generally substantial stores of fine particulate organic matter (FPOM), and very large accumulations of OM can become anoxic. Recent research has demonstrated that even these accumulations contribute to small stream food webs via methanotropic microbes (Kohzu *et al.*, 2004), although it is too early to tell yet whether this is a distinguishing feature of small, headwater streams.

The food webs, are therefore, driven by detritus. The CPOM may be consumed directly by invertebrates called shredders, while others rely on microbial biofilms or FPOM derived from the CPOM. Most of the detritivores are small macroinvertebrates (body length typically > 20 mm) and meiofauna (> 250 μm). In Tasmania, the largest invertebrates that might be found in such small streams are some smaller specimens of crayfish (*Astacopsis* spp.) but they are hardly ubiquitous in these far upstream reaches of the network (Davies *et al.*, 2005b).

7.2.3. In-stream vertebrates are scarce

Vertebrates are scarce. Fish are typically absent, owing to the shallowness of the water and plentiful barriers to dispersal from downstream (Richardson and Danehy, 2007). Notwithstanding this, much North American research has focussed on salmonids, and has usually been conducted in larger streams further down in the catchment and so these studies are not really in "headwater streams" as defined here (Richardson and Danehy, 2007). However in western North America, the top predator niche in some truly small headwater streams has been occupied by salamanders, and they are probably using these habitats as a refuge from fish predation lower down in the catchment (Richardson and Danehy, 2007). In Tasmania, the herpetofauna (especially frogs) remains poorly characterised, and research on frogs has tended, as with most taxa, to focus on larger water bodies further downstream in catchments: (Lauck et al., 2005b, Lauck et al., 2005a). Platypus, however, are unique to Australia, and they have been found in unlogged small streams in north-east Tasmania, and the suggestion is that these are dispersing, young individuals rather than being long-term residents in these streams (Koch *et al.*, 2006).

7.3. Biodiversity values

7.3.1. Macroinvertebrates

Headwater streams are often cited in the literature as harbouring unique species found nowhere else in a catchment (Meyer *et al.*, 2007), but much of this literature refers to streams much larger than the small headwater stream network encompassed by the notion of the Class 4 streams as defined under the *Forest Practices Code* (Forest Practices Board, 2000) (Clarke et al., in review). Very little has been published internationally comparing the biodiversity of a range of groups in small headwater streams with that of downstream reaches, and some suggest that maximal biodiversity in river systems is found in the middle reaches, with the headwater network being relatively depauperate (Heino et al., 2008). Nevertheless when several headwater streams are compared in the same region, there can be considerable differences between them (Heino et al., 2005a) implying that there can be much species "turnover" (or high β-diversity) as one moves across the streams at the head of a catchment. Again, however, many of the published empirical results relate to streams draining larger catchments than those considered in this section (Clarke et al., in review, Barmuta et al. in prep.).

Our current state of knowledge in Tasmanian small headwater streams provides two potentially conflicting insights. The first is that much of the macroinvertebrate fauna of small streams seems to consist of a subset of the fauna found further downstream. The other is the high degree of local endemism found in some groups of freshwater or freshwater-dependent groups of invertebrates that have been investigated by taxonomic specialists.

The first, based on extensive surveys across more than a dozen logged and unlogged streams in the Southern Forests suggests that the macroinvertebrate fauna is dominated by aquatic oligochaetes and dipteran fly larvae, with only sporadic occurrences of larger amphipods, ephemeropterans, plecopterans, trichopterans and coleopterans (J.P.R. Gooderham, University of Tasmania & DPIW, unpublished data). There is much variation between even adjacent streams in the same catchment in terms of abundance and composition, and community changes that could be attributed to logging activities vary in their strength (Watson, 2005, Davies et al., 2005a).

The second insight is the prevalence of narrow-range taxa (sensu Ponder and Colgan, 2002) in some parts of Tasmania. The principal in-stream example comprises small, fully aquatic hydrobiid snails, especially in the genus *Beddomeia* (Ponder and Colgan, 2002), and their distribution and potential responses to forestry activities is currently being investigated (K. Richards, Forest Practices Authority & University of Tasmania). Burrowing crayfish provide a further example of species with very narrow geographic ranges, with adjacent catchments supporting different species (Hansen and Richardson, 2006, Hansen and Richardson, 2002, Whiting et al., 2000, Horwitz, 1988). While these crayfish are not fully aquatic (although they do depend on water to survive in their burrows), and many species are found in catchments larger than those encompassed by the Class 4 definition, the potential for narrow-range, endemic taxa in Tasmania is clear.

The biogeographic processes causing narrow-range endemism are only partially understood for Tasmanian freshwater taxa. Thus far, northern Tasmanian catchments seem to support more of these taxa than eastern and south-eastern systems, and better understanding of the recent glacial history of Tasmania's catchments may help identify those which are more likely to support endemic species. For example, it could be that the north-draining rivers of Tasmania provide a downstream refuge on the exposed Bassian Plain during glacial maxima that is unavailable to rivers draining east (or presumably south and west).

However, a precautionary approach to appraising the biodiversity of small headwater systems is preferable until further, focussed research is undertaken. Much of the invertebrate biodiversity of small headwaters is uncharismatic, cryptic and requires specialised skills for positive identification. Aquatic oligochaetes, dipteran larvae (especially in the Chironomidae, or non-biting midges) and most of the meiofaunal taxa (e.g. Copepoda, Ostracoda and other microcrustacea, Rotifera, Nematoda, Tardigrada) require compound light microscopy or even dissection or sectioning for identification to species level. This is a substantial taxonomic impediment to quantifying biodiversity of small headwater streams, and it is unlikely to be resolved within the next two or three decades. Moreover, even within the more macroscopic species, there is likely to be much cryptic diversity that will emerge from finer scale morphological and molecular examination of the are currently described taxa.

7.3.2. Macrofauna

This *ad hoc* term is taken to mean mammals, fish, amphibians, and large invertebrates (with adult body lengths > 20 cm). In Tasmania most headwater systems appear to be fish-free, although some of the larger streams that get classified as "Class 4" and those that discharge directly into much larger streams may harbour the extreme upstream ends of the distribution of some native fish that are able to cope with the natural barriers usually encountered lower in the catchment.

Similar generalizations hold for freshwater crayfish (*Astacopsis* spp.), but most information about habitat requirements have focussed on the listed *Astacopsis gouldi*. It is scarce in small headwater systems, and is only present in these streams when they become steep enough to develop distinct pools (Davies and Cook, 2004, Davies et al., 2005b). Geological contact zones (areas where the probability is higher of contributions by groundwater owing to contacts between different geologies) can be used as a proxy, in concert with slope information, to map the headwater streams that are most likely to harbour this species (Davies *et al.*, 2007).

7.4. Forestry impacts and management

7.4.1. Barriers, roading and harvesting

The majority of small headwater streams in Tasmania are likely to be naturally free of fish and so road-associated culverts are only likely to pose problems to fish dispersal if the stream is in a low-relief landscape.

As a result, the chief impacts from roading are likely to accrue from sediment delivery to the stream and, this linked to the harvesting regime and associated catchment disturbances. Croke and Hairsine (2006) argue that better management of sediment inputs could be achieved by more careful consideration of “source areas” and delivery pathways in a catchment and affording those more protection (Figure 4).

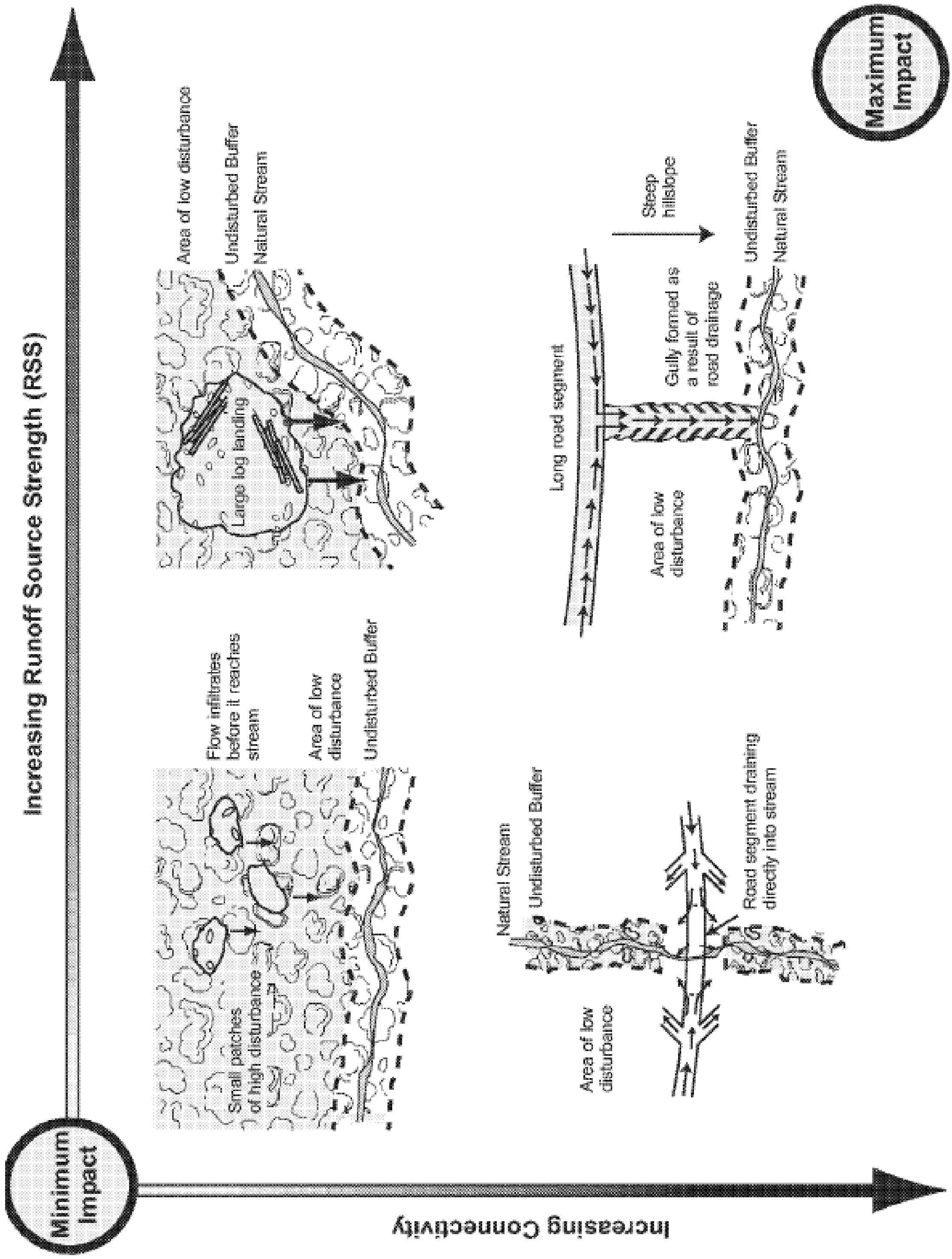

Figure 4 A conceptual framework showing the relationship between the strength of sources of runoff (RSS) and connectivity in a catchment. (From (Croke and Hairsine, 2006).)

"Source areas", loosely, refer to those (terrestrial) parts of the catchment that tend to yield most of the water that flows into a stream. Catchments are not uniform in their behaviour as they get wetter during a prolonged period of precipitation, and so some parts of the catchment are more likely than others to generate overland flow. Croke and Hairsine (2006) argue that such source areas vary in their "strength" or potential to yield runoff, so the first axis of their conceptual model represents runoff source strength (RSS). How directly these source areas connect with streams is termed "connectivity", thus yielding the second axis of their conceptual model (Figure 4).

Catchments and silvicultural systems will differ in their RSS potential and the connectivity with stream network. CBS systems on relatively impermeable soils would have much greater RSS than shelterwood harvests for example, but systems that required frequent vehicular access might require more attention to managing flow paths and connectivity than one where regeneration was allowed to progress with little other intervention during the rotation.

Croke and Hairsine (2006) conclude by tabulating the measures that could be implemented to manage RSS and connectivity on forest coupes. Rather than advocate single, "one-size-fits-all" prescriptions, their scheme raises the prospect of re-casting the management of sediment inputs in a risk-based framework: if a particular harvesting regime and soil type result in high RSS and potential for connectivity, then more stringent prescriptions would be needed than for a catchment on soil types that yield weaker RSS, for example.

This mirrors the current approach to a risk-based method for determining MEZ and streamside reserve prescriptions for Class 4 streams (McIntosh and Laffan, 2005)[8]. A combination of managing RSS and the pathways leading from source areas is likely to be a more effective means of reducing forestry-related impacts than solely relying on streamside reserves and elegant road crossings. The challenge will be to identify these features prior to harvest, and provide a decision support system that is as easy to implement as the current Class 4 guidelines. While Croke and Hairsine (2006) emphasise further research needed to understand the heterogeneity of sediment yields within catchments, they point to spatially-explicit, fine-scale GIS based tools to evaluate risks at the level of individual coupes or management units.

Table 4 Croke and Hairsine's (2006) summary of measures to manage RSS and connectivity in forestry.

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[8] See also the FPA Class 4 Guidelines on: http://www.fpa.tas.gov.au/index.php?id=67&tx_avotherresources_pi1[action]=ResByCat&tx_avotherresources_pi1[cat]=36

Management of runoff source strength	Management of sediment delivery path way
Unsealed roads	Unsealed roads
Drainage spacing	Drainage spacing
Road drains need to be constructed and maintained at spacings that minimize the sheet and rill erosion on the road	The spacing of road drains should also consider the design criteria of preventing gulling at the road drainage outfall and the minimization of the delivery of diffuse overland flow to the stream
Surfacing	Drainage position
Resurfacing of roads should attempt to minimize the availability of fresh fine sediment for subsequent erosion	Road drains should not discharge into or near drainage lines. They should discharge onto planar or diverging slopes
Position	Road position
Wherever possible roads should be positioned on ridge tops or near to ridge tops to prevent capture of runoff from the hillslope above	Roads should be positioned to maximize the length of the flow path of overland flow leaving the road
Traffic management including closure	Stream crossings
Where practical, forestry roads should be restricted or closed to traffic, which disturb the road surface thereby generating more available sediment	Stream crossings should be constructed to minimize or eliminate the area road draining directly into a stream
Harvest tracks and other compacted areas	Harvest tracks and other compacted areas
Banks	Location
Compacted tracks and landings should be segmented into small units by banks to disperse runoff water	Harvest tracks should be laid out from ridge top log landings
Location including landings	Banks
Log landings should be located on ridge tops to minimize the runoff capture from upslope	Post logging closure of tracks should ensure runoff is dispersed in adjacent harvest areas and buffers
General harvest area	General harvest area
Minimizing disturbance	Selective logging
Disturbance of the soil during logging operations should be minimized so that the majority of the soil surface remains protected by litter and vegetation cover	Selective logging should be used wherever possible to maximize the flow path length in low disturbance areas of runoff from the harvest area
Buffer zones	Buffer zones
Disturbance	Extent
Buffer zones should be subject to no or minimal disturbance during the logging operations	Buffer zones should protect all drainage lines including ephemeral drainage lines
Extent	Width
Buffer zones should extend to protect all drainage lines including ephemeral drainage lines	Buffers should be of adequate width so that they are able to filter inflowing overland flow (approximately 20 m either side of all drainage lines)

7.4.2. Harvesting and regeneration

Table 5 summarises the broad responses by small headwater streams to forestry in Tasmania and in temperate forest overseas. Much of the published literature focuses on short-term impacts and responses, because much of the research on biodiversity in these streams only started in the last decade, and many jurisdictions have implemented new forest practices over a similar time span which make comparisons with forests regenerated before these altered practices difficult.

Table 5 Responses of Features of small headwater streams to forestry activities.

Feature	Changes attributed to harvesting	Responses	References
Low hydraulic power	Higher peak discharges; flashier flows.	Increased scour of previously benign habitats. Probably more increased exports of OM and nutrients, longer nutrient spiralling paths.	
Geomorphology	Increased incision or Channelization of stream bed. In extreme cases, increased erosion of bed and banks; bank failure; increased probability of debris flows.	Change of proportional representation of habitats. In extreme cases, dramatic alteration of in-stream habitat.	
Flow paths and temperature	Routing of overland flows into streams via roads, drainage lines	Sedimentation; interstices filled with fine (inorganic?) sediment. Altered microbial functioning.	
Flow seasonality	Peak winter high flows may be higher; unclear whether, low flows or dry periods are prolonged.	Elevated delivery of sediment during peak discharges	(Clapcott, 2007)
Disturbance regimes	Vulnerable to low flows, landslides or debris flows	Habitat loss and isolation	(Swanson *et al.*, 1998)
Recolonization pathways	In-stream movements may be impeded by slash, fallen wood; riparian changes may alter flight patterns.	Recruitment loss or failure of fish species; reduced dispersal	(Fagan, 2002)
Aquatic-terrestrial linkages	Reduced habitat for terrestrial fauna	Energy flows to terrestrial compartment	(Wipfli, 2005) (Wipfli et al., 2007)

	immediately after harvest; less allochthonous CPOM. Regrowth/regeneration in riparian zone may favour different mix of plant species	may be reduced; possible reduction of secondary production delivered to downstream reaches. Changed nutrient regimes if recolonising spp. fix nitrogen	(England and Rosemond, 2004) (Wipfli and Musslewhite, 2004)
Aquatic vertebrates	Temperature effects on spawning or recruitment of fish & amphibians; effects may carry downstream. Changes to secondary production; sometimes short-term increase immediately after harvest followed by declines later.	Decreased fish or amphibian abundance. Platypus absent from previously harvested reaches.	(Richardson *et al.*, 2005) (Koch *et al.*, 2006)
Canopy closure	Light reaching stream bed increases initially followed by decline as canopy closes over. Initial decrease of leafy CPOM, but accidental inputs of LWD may result from harvesting or increased wind throw during regeneration and regrowth.	Initially, localised benthic algal blooms with increased use of these resources in food webs. Initial decline in retention of leafy CPOM or an increase if LWD increases retentiveness. Less CPOM for a period prior to re-establishment of larger woody riparian species	(Clapcott, 2007) (Danehy *et al.*, 2007) (Watson, 2005) (Davies *et al.*, 2005a)

The most comprehensive data for Tasmanian headwater streams that is available is from research conducted as part of an ARC Linkage grant with the School of Zoology, University of Tasmania, Forest Practices Authority and Freshwater Systems Pty Ltd. Full details are supplied in Clapcott (2007) and Watson (2005).

In terms of ecosystem processes, CBS harvesting in tall, wet eucalypt forests on dolerite/sedimentary complexes results three classes of response over the period examined (up to 14 years after harvest) using space-for-time surveys (Figure 5). Measures of community respiration show little change in response to logging, while measures of primary production show an initial marked increase, with maximal rates around 5 – 7 years after harvest and then a decline towards pre-harvest levels by 14 years. The permanent changes after harvesting are in the the heterotophic processes (bacterial carbon production, decompostion potential) that continue to change 14 years after harvesting. There is some indication that feeding activity by large-particle detritvores also continues to be impaired for this period of time.

These changes suggest some form of hysteresis in the *heterotrophic* metabolism of these streams, and this is reflected in the data suggesting permanent physical changes to their configuration of in-stream habitats. In dolerite/sedimentary complex streams, the number of in-stream habitats

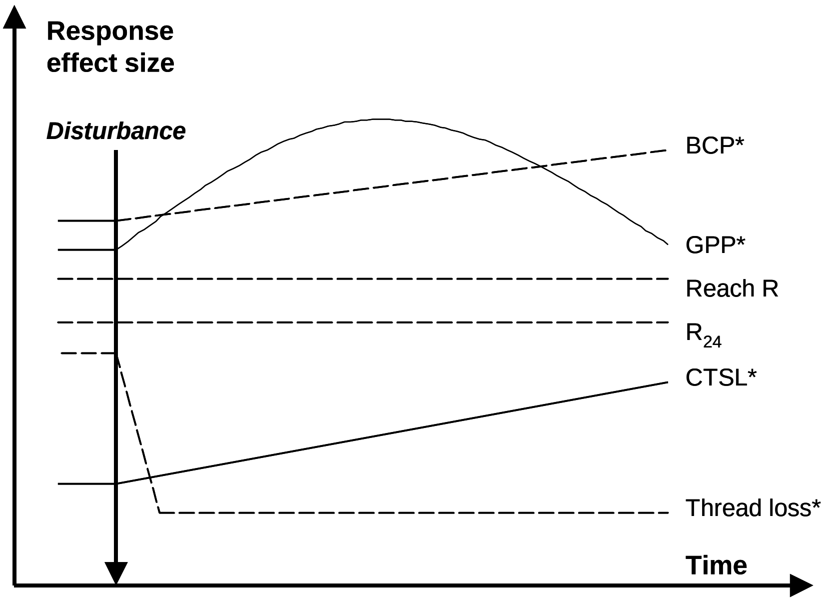

Figure 5 Summary of trajectories observed for metabolic variables in small headwater streams up to 15 years after the logging of catchment vegetation. Solid lines indicate variables where a significant regression with time was observed, and * indicates variables where a significant difference between logged and unlogged streams was observed. BCP = Bacterial Carbon Production; GPP = Gross Primary Production; Reach R = Reach-scale Community Respiriation; R_{24} = patch-scale Community Respiration; CTSL = Cotton Tensile Strength Loss (the measure of Cellulose Decomposition Potential); Thread loss = loss of threads from cotton strips used for CTSL, and is a coarse index of feeding activity by large-particle detritivores

increased immediately after logging, with more fine-sediment habitats being recorded, and the proportion of the stream bed flowing "underground" was markedly reduced. The latter probably resulted from a combination of increased stream erosive competence caused by the intial increases in runoff, and the removal of the organic "rooves" over the top of segments of streams by the regeneration burns. (Much of the covering of these streams consists of decomposing logs and other woody material that has fallen across and spanned the stream beds.)

There are three major caveats. First, the data are very variable, suggesting that, after logging, some of the response variables increase in dispersion (loosely "variance") as well as or instead of changes in location (loosely "mean" or "median"). These data were very sparse for some year-classes after harvest, and so there may be important changes in some of the "non significant" variables that have been missed in this survey. Second, the length of successional sequence covered by this survey remains short (14 years) relative to the potential rotation time for this form of native forest logging. Accordingly, what appears to be hysteresis in this survey may alter as the streams continue to change in response to the initial increases in runoff and sediment delivery after the regeneration burn, and as riparian and catchment vegetation continues to develop as part of the forest's regeneration. Third, the broad patterns found here may not be reflected in all geological contexts. At Ben Nevis, for example, harvesting resulted in a marked and prolonged increase in the representation of erosional habitats rather than depositional habitats, although this pattern is possibly confounded with the lack of any streamside protection at the time that these streams were logged (Davies et al., 2005a).

7.5. Machinery Exclusion Zones and beyond

Many forestry jurisdictions provide little or no protection for small headwater streams nor for intermittent watercourses. Tasmania was probably among the first to afford some protection to headwater streams, and this is becoming more prevalent in other wet, forested areas. Olson et al. (2007) collated over a dozen different prescriptions from the Pacific Northwest. These include "no-cut buffers" ranging in width from 0 to >100 m, and various "management zones" spanning a similar range of widths. In some jurisdictions, no-cut buffers are delineated closest to the stream, and a management zone is added to the upslope margin of the buffer. The motivations for these protection zones are various, but the widest zones are generally reserved for fish-bearing streams. In this region "fish = salmon", and there is also a large literature on temperature changes resulting from forestry, and prescriptions on covering streams with slash to ameliorate temperature rises have been used in the past in clearcut harvest systems (Olson *et al.*, 2007).

The diverse amphibian fauna of this region has recently broadened the focus of stream protection. This group of vertebrates uses stream corridors for dispersal, but some species also can move across watersheds (Suzuki and Olson, 2007). Thus connectivity between metapopulations can only be maintained by having purely terrestrial corridors or stepping-stones (also called "leave islands") across the catchments (Figure 6, parts e, f & g). Olson et al. (2007) term this the "spaghetti and meatballs" approach. Olson et al. (2007) and other authors proposing more complex matrix-like networks of habitat emphasise that not all of these buffers and islands need be unharvested, and that a mix of intensities of land use across managed forests would be the most defensible approach (Suzuki and Olson, 2007).

However, all these alternatives remain conceptual models. More conventional approaches attempting to document in-stream responses to varying buffer widths are being tested in field situations in several places in North America (cited in Richardson and Danehy, 2007). Most of these studies are relatively young, and the results really only bear on the short-term effects immediately after harvest, and some comparisons between buffered and unbuffered streams find little change in macroinvertebrate communities in response to clearfell-style harvesting (Moldenke and Ver Linden, 2007). Generally, Richardson and Danehy (2007) report that the responses are variable (Danehy et al., 2007, Olson and Rugger, 2007), and that no buffer prescription is likely to completely protect in-stream biota. As has been noted more generally for perturbations to running waters, there is no such thing as "no effect": the question then becomes "how much change is acceptable?" (Downes et al., 2002).

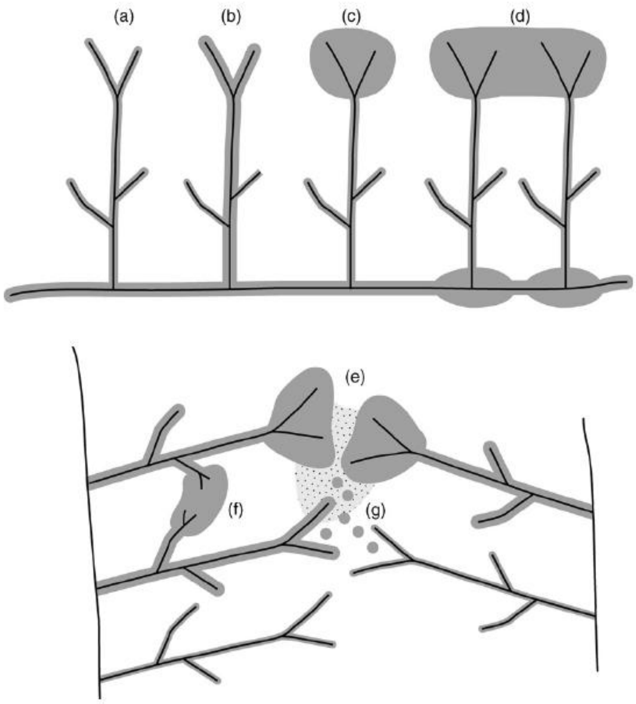

Figure 6 Various configurations of "buffers" around headwater and downstream sections (from Figure 3 of Olson *et al.*, 2007) for protecting amphibians in western North American headwater networks. Key: (a) narrow buffer zone might be used to protect water quality and some instream habitat components where headwater amphibian occurrences or habitat quality are low; (b) wider buffer zone contributes to retention of instream and riparian habitat conditions and some biota; (c) patch reserves at headwaters to protect endemic species and functions contributing to downstream habitats, and to provide connectivity between joined headwater channels; (d and f) patch reserves can provide connectivity across ridgelines to adjacent drainages, and can be placed downstream to provide enhanced riparian habitat protection such as at tributary junctions; (e and g) partial harvest (shaded area) and/or leave islands (circles) may be used to provide connectivity functions between catchments.

In Tasmania the 10 m Machinery Exclusion Zone (MEZ) for Class 4 streams can be upgraded in areas that are prone to higher risk of erosion (Forest Practices Board, 2004)[9], and this additional "safety factor" applied to headwater streams is unusual internationally (Richardson and Danehy, 2007). Because increased delivery of sediments to streams is an important impact on in-stream flora and fauna, this greater level of protection afforded to such headwater streams is likely to be beneficial.

However, the MEZ or even extended streamside buffers or reserves are unlikely to afford complete protection to in-stream biota. Streamside buffers in general are usually breached by small drainage lines that can deliver

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[9] Note also that the presence of threatened species or other species of conservation interest can increase the protection afforded to Class 4 streams.

sediments and sorbed pollutants and dissolved pollutants during times of overland flow (Norris, 1993). Even in unlogged catchments draining periglacial doleritic material in Tasmania, the headwater system is often a complex network of streams and tributary “macropores” covered by fallen logs and living tree roots, and the streams themselves may flow for many metres through such tunnels and pores (J.P.R. Gooderham, University of Tasmania, unpublished data).

When harvested, buffers may be breached by fallen trees and logs more frequently than would have occurred prior to harvest, and this may prevail for many years because the remaining riparian trees are more exposed to wind throw owing to the lower height of the surrounding vegetation. These windthrows may result in large disruptions to the stream bed habitats (e.g. exposure of rootballs, creation of deep holes in the stream bed), expose previously subterranean sections to sunlight, and even alter the course of the stream (J. Clapcott, J.P.R. Gooderham, University of Tasmania, personal observations). More immediately, MEZs and stream protection reserves are vulnerable to fire during regeneration burns, and such fires may temporarily remove the complex surface layer of terrestrial detritus that would slow and intercept overland flow and transport of sediment.

Finally, roading and snigging often increase the delivery of sediment to the streams (Section 6.2), even in the best-designed and implemented systems of road management.

7.6. Exotic species

While not exclusively an impact derived from forestry activities, the potential for other agents to affect headwater biodiversity needs to be acknowledged, although it may be difficult to control these agents. For headwater streams in Tasmania the two major potential agents for altering biodiversity are lyrebirds and the translocation of introduced fish.

After establishing long-term study sites in the Warra LTER, our research group noted increased activity of the introduced lyrebird in some of these streams (J.E. Clapcott, J.P.R. Gooderham & L.A. Barmuta, University of Tasmania, personal observation). This resulted in localised, but often very substantial scratching and digging in stream beds and adjacent riparian areas, and we speculate that the wetter areas close to Class 4 streams may be targeted by lyrebirds during droughts. Lyrebird activity may disrupt flow pathways, rearrange in-stream and riparian habitat, overturn moss-covered rocks and large-woody debris (that would not otherwise be overturned by these small streams) and alter the morphology of streams. The extent to which their activities will threaten in-stream biodiversity remains to be investigated.

The illegal translocation of fish (chiefly salmonids) by fishing “enthusiasts” remains a threat to those headwater streams that are capable of sustaining fish. This is more likely to be a major issue for larger streams (i.e. Class 3 and larger), and so was treated more fully in Section 6.2.

The other issue with exotic species that is more closely allied with forestry is changes to the riparian vegetation and associated inputs to small streams. In many older harvesting and conversion regimes, there was little or no protection afforded to headwater streams. Consequently, a range of riparian changes are obvious in the Tasmanian landscape. Older pine plantations have resulted in complete replacement of the riparian vegetation by *Pinus radiata*. Conversion of other land uses to plantations of various types may result in non-indigenous eucalypt or other species becoming prominent in the riparian zone. Increased access and roading activities also provides opportunities for introducing a variety of exotic species, of which blackberries (*Rubus* spp.) are a prominent example. Non-native riparian plants can affect the timing and quantity of CPOM delivered to the in-stream fauna, and *Pinus radiata* litter, in particular, is likely to be much poorer substrate for in-stream food webs than native eucalypt species.

7.7. Conclusions on protecting biodiversity in headwaters

Overall, it would be rash to conclude that Tasmania’s headwater streams supported little unique biodiversity. There may be regional patterns in the potential for endemism that need further, targeted biogeographic research to validate. If such regionalism can be substantiated, it would support a more diverse approach to prescriptions to protect such streams.

In the shorter term, however, it is clear that there are some effects on the ecosystem processes that support this biodiversity. In the tall, wet forests of the around Warra LTER subject to CBS harvesting systems, these effects last for at least 14 years, and further research is clearly needed over the longer term in this and other forest types in Tasmania to better estimate the likely return times of these processes. Accordingly, some precautionary principles need to be formulated.

The obvious option is to increase the width of MEZ or even to upgrade their status to that of the stream side reserves (SSRs) on larger streams. In steep terrain, increasing the width of MEZs would likely render any timber harvesting infeasible. Likewise increasing their status would be problematic in harvesting systems where burning was a necessary part of the management. Preventing burns from escaping into MEZs would be very difficult. Finally, no MEZ or SSR is likely to be unbreached by obscure, temporary drainage lines and other features that would compromise their "buffering ability" during times of increased overland flow.

To better conserve the biodiversity in headwater streams, I suggest that a proportion of the headwaters of a catchment be temporarily reserved from forestry operations for the duration of a mangement cycle or rotation. This proportion of the network should be connected with SSRs at its downstream extremity to maintain longitudinal connectivity. It would be preferable also to ensure that its highest altitudinal segment conneced with habitat strips, biodiversity spines, adjacent formal reserve or a similarly reserved portion of the headwaters of the adjacent catchment to ensure lateral connections for the dispersal of those species that use mature forest in catchment headwaters.

As with some other informal reserves in the Tasmanian forest practices system, these reserved portions of the headwaters need not be a permanent feature of the landscape. As previously harvested headwaters in the same catchment become mature and connect with other mature components of the "habitat matrix", then this initially reserved area could then be harvested. Figure 6 provides some alternative configurations, and further research is obviously needed to evaluate these alternatives.

Overall, such a scheme is likely to spread the risk of disconnecting headwater streams from adjacent terrestrial habitats upon which some of their species might depend. Similarly, the longitudinal connections are maintained somewhere within the catchment to afford access to headwaters by dowstream species which depend on headwater habitats for some requirements.

The precise quantitative proportion of a catchment that needs to be temporarily reserved is difficult to specify given that there are no data anywhere that evaluate this approach in terms of its value to biodiversity. Currently, the Threatened Fauna Advisor of the Forest Practices Authority provides guidelines to protect streamflow for Swan Galaxias based on Vertessy's (1999) review. This might provide a first approximation of proportions of catchments that could be harvested, although I note that modelling by Meggs et al. (2004) query the usefulness of this approach for this particular species in regenerating forest.

8. Other Tasmanian aquatic biodiversity issues and forestry

8.1. Unusual components of the fauna

Several biodiversity issues related to freshwaters and forestry are prominent in, and sometimes unique to Tasmania. Endemic species with narrow geographic ranges, or “narrow range endemics”, are a prominent feature of the north- and west-draining catchments of Tasmania (and of catchments in the south-west, most of which are excluded from forestry). These are exemplified by the hydrobiid snails in the genera *Beddomeia* and *Phrantella* in Figure 7.

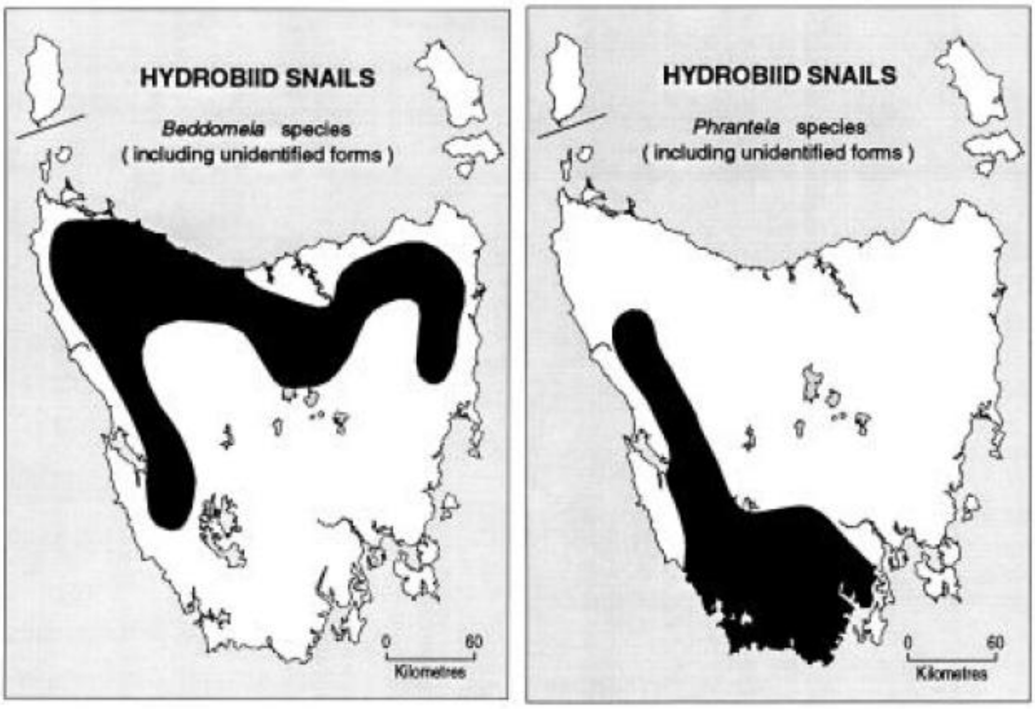

Figure 7 Generalized distribution maps for *Beddomeia* and *Phrantella* species in Tasmania. (Source: Fauna Value Database, Forest Practices Authority of Tasmania, http://www.fpa.tas.gov.au/index.php?id=82).

Burrowing crayfish (e.g. *Engaeus* spp.) are not, strictly, stream or pond dwellers, but their reliance on heavily saturated soils usually means that drainage depressions, seeps and other fine-scale habitats associated with streamlines and wetlands need to be protected to ensure the persistence of these species. Again, these species are mostly confined to north- and west-draining catchments (Hansen and Richardson, 2002, Hansen and Richardson, 2006, Horwitz, 1988, Horwitz, 1990) . These coincident biogeographic patterns in these poorly-dispersing species that depend on aquatic features of the landscape suggest that common processes have caused the high degree on endemism in the north and west of Tasmania, and suggest that a regional approach to quantifying the potential for narrow-range endemics might be worth pursuing in amending the Forest Practices Code.

Some of the **endemic freshwater fish** also have restricted distributions: some are confined to limited catchments in the Central Highlands, others in the far north-east and others to some eastern catchments. Some of the more fragmented distributions doubtless reflect the impacts of habitat destruction and the introduction of piscivorous recreational fish such as brown and rainbow trout (Hardie et al., 2006).

Cave-dwelling fauna consists of a variety of aquatic and non-aquatic invertebrates. They fall within the ambit of aquatic biodiversity since the vast majority of the cave systems supporting these species are in limestone and water moving through these formations create and maintain the cave habitats that hose these species.

There are, of course, other of special conservation interest including those listed under threatened species legislations (e.g. Giant Freshwater Lobster, *Astacopsis gouldi*). The Forest Practices Authority maintains a database of all such species and species groups (currently called the Fauna Value Database, http://www.fpa.tas.gov.au/index.php?id=82), with specific variations to standard prescriptions to provide additional protection to the habitat elements required by these species. For example, endemic hydrobiid snails in Class 4 streams activate more stringent streamside reserves than the standard MEZ specified by the Forest Practices Code. Further research is necessary to evaluate and critique the provisions that have been recommended, and some of this is currently in progress (e.g. hydrobiid snails by K. Richards).

8.2. Taxonomic impediments

Tasmania shares patchy and often poor taxonomic resolution of its aquatic biota with many parts of the world. Managing in the face of incomplete taxonomy and therefore knowledge of biodiversity risks inadvertently loosing species before they even become known to science (Kim and Byrne, 2006). The use of surrogates and umbrella species to substitute for incomplete knowledge has a very mixed record (Heino et al., 2007b and references therein). Alternative approaches that use relationships between various metrics of diversity show some promise (Heino et al., 2007a, Heino et al., 2005b), although identification to voucher species level is still needed across broad geographic regions to validate these techniques. Exploiting and improving our current understanding of historical biogeography of Tasmania holds much promise. Horwitz (1988) and Hansen and Richardson (2002) argue convincingly for historical processes determining the distributions of parastacid crayfish, and such information could be useful in prioritizing further research into biodiversity hotspots within Tasmania. Historical biogeography can also be useful in identifying glacial refugia (Kirkpatrick and Fowler, 1998), and such information will be crucial in managing for climate change in the medium to long-term.

8.3. Issues related to plantations and conversion

There is little reasearch about the impacts of forestry plantations on aquatic biodiversity that is relevant to current Tasmanian forest practices, and this is a major gap in our knowledge that needs to be accorded a higher priority. The limited overseas research in temperate zones does point to some mechanisms which may prevail in local streams subject to plantation conversion, and continuing plantation management.

Older-style plantation establishment rarely left in-tact native vegetation in the riparian zones, and this has altered the nature of litter inputs (timing and quantity), and the accession of dead timber to the stream (Broadmeadow and Nisbet, 2004). Some New Zealand research has shown that introduced *Pinus* wood can contribute substantially to the diets of some stream invertebrates in pumice-bedded streams (Collier and Halliday, 2000), while a more extensive comparison between *Pinus* and native wood revealed no differences between the timbers in terms of the communities that developed on them (Collier et al., 2004). Plantation management has often included removal of in-stream timber, with deleterious impacts on fish that rely on this habitat for cover or spawining (Broadmeadow and Nisbet, 2004). Less in-stream wood also reduces the retentiveness of a stream, leading to lower densisties of detritivores, and artificially “re-snagging” streams in plantations has resulted in substantial recovery of these attributes (e.g. Pretty and Dobson, 2004).

The potential ramifications for aquatic detritivores of chemical differences between native leaf litter and introduced plantation species have received more attention. In Victoria, O'Keefe and Lake (1987) found that eucalypt litter broke down faster than that of *Pinus radiata*, although they also found that native accacia phylodes were processed as slowly as the pine needles. The tables are turned somewhat overseas, where introduced eucalypts are viewed as the villains: leachates from eucalypt litter in small stream pools killed the dominant shredder in one Portugese study (Canhoto and Laranjeira, 2007), and other studies in this region have shown the introduced eucalypts provide inferior food for at least some detritivores (Canhoto and Graça, 1995).

Elsewhere, effects of forestry plantations on in-stream biota are documented in environmental contexts that differ from those prevailing in Tasmania. Of the temperate studies, those in north-western Europe are confounded with reforestation and other efforts to control acidic deposition (Giller and O'Halloran, 2004), and studies generally find a weaker relationship with riparian vegetation conditions than with catchment acidification (e.g. Ormerod et al., 1993). There are many initiatives to restore a more "native"riparian zone in many plantations, but the evidence for the beneficial in-stream effects is patchy, and more empirical research is called for (Giller and O'Halloran, 2004)(Broadmeadow and Nisbet, 2004)

Given the likely scenarios for Tasmanian forestry activities concerning plantation management and conversion to plantations, the following research topics are probably the highest priorities for aquatic biodiversity:

1. Research into the opportunities for restoring in-stream habitat and riparian zones throught the conversion of previously cleared land to plantations, especially of eucalypt species. This would include research directed at estimating appropriate loads of in-stream woody debris, and the rate at which they would need to accumulate as these attributes would likely vary across the different forest types in Tasmania.

2. Evaluations of the aquatic biodiversity values that are currently conserved in exotic plantations. As in New Zealand, even pine plantations with no native riparian buffer may still provide some biodiversity benefits, but the degree to which these deviate from intact native needs to be established if defensible restoration goals are to be framed.

3. Examination of the potential for the amelioration of riparian conditions in exotic plantations established prior the Forest Practices Code. Such plantations (chiefly of radiata pine) rarely have any native riparian vegetation remaining, and harvesting these plantations presents an opportunity to re-establish native riparian species within these catchments.

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