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Biodiversity management approaches for stream–riparian areas: Perspectives for Pacific Northwest headwater forests, microclimates, and amphibians

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Abstract

Stream–riparian areas represent a nexus of biodiversity, with disproportionate numbers of species tied to and interacting within this key habitat. New research in Pacific Northwest headwater forests, especially the characterization of microclimates and amphibian distributions, is expanding our perspective of riparian zones, and suggests the need for alternative designs to manage stream–riparian zones and their adjacent uplands. High biodiversity in riparian areas can be attributed to cool moist conditions, high productivity and complex habitat. All 47 northwestern amphibian species have stream–riparian associations, with a third being obligate forms to general stream–riparian areas, and a quarter with life histories reliant on headwater landscapes in particular. Recent recognition that stream-breeding amphibians can disperse hundreds of meters into uplands implies that connectivity among neighboring drainages may be important to their population structures and dynamics. Microclimate studies substantiate a ‘‘stream effect’’ of cool moist conditions permeating upslope into warmer, drier forests. We review forest management approaches relative to headwater riparian areas in the U.S. Pacific Northwest, and we propose scenarios designed to retain all habitats used by amphibians with complex life histories. These include a mix of riparian and upslope management approaches to address the breeding, foraging, overwintering, and dispersal functions of these animals. We speculate that the stream microclimate effect can partly counterbalance edge effects imposed by upslope forest disturbances, hence appropriately sized and managed riparian buffers can protect suitable microclimates at streams and within riparian forests. We propose one approach that focuses habitat conservation in headwater areas – where present management allows extensive logging – on sensitive target species, such as tailed frogs and torrent salamanders that often occur patchily. Assuming both high patchiness and some concordance among the distribution of sensitive species, protecting areas with higher abundances of these animals could justify less protection of currently unoccupied or low-density habitats, where more intensive forest management for timber production could occur. Also, we outline an approach that protects juxtaposed headwater patches, retaining connectivity among sub-drainages using a 6th-field watershed spatial scale for assuring well-distributed protected areas across forested landscapes. However, research is needed to test this approach and to determine whether it is sufficient to buffer downstream water quality and habitat from impacts of headwater management. Offering too-sparse protection everywhere is likely insufficient to conserve headwater habitats and biodiversity, while our alternative targeted protection of selected headwaters does not bind the entire forest landscape into a biodiversity reserve.

Keywords: Riparian buffers; Riparian patch reserves; Amphibians; Stream and riparian microclimates; Connectivity; Riparian forest management approaches

1. Introduction

The values provided by streams and their riparian zones within forested landscapes continue to be a focus of high concern and contention. New science has redefined the resources of interest for production, retention or restoration in these areas as well as the scope of threats to these systems. Simultaneously, new management approaches have underscored the diverse priorities among land managers. These issues are especially acute in the Pacific Northwest forests of North America, where neighboring lands have diverse forms of stream–riparian protection, ranging from none to entire

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drainages (e.g., Sheridan and Olson, 2003; USDA and USDI, 1994; Sedell et al., 1994). This diversity is reflective of the different resources of importance for management, and land ownership varying from individual private woodlot owners to state and federal resource agencies. Fueling the debate of how to approach riparian management zones, the role of large reserves across the forested landscape as the cornerstone for maintaining forest biodiversity and ecological integrity is being challenged. Coarse-scale reserve networks often fail to capture rare species (Thomas et al., 1993; Lawler et al., 2003; Molina et al., 2006; Olson, 2006). Similarly, ecological processes, both biotic and abiotic, function across all scales (Lowe et al., 2006), and many likely should be retained across entire landscapes, not just inside reserve areas.

The value of riparian management as a finer-scale reserve network may have substantial benefits as habitat or dispersal corridors (e.g., Rosenberg et al., 1997) for a suite of northwestern species (USDA and USDI, 1996a,b), especially because some level of riparian protection exists within every subdrainage (Pollock et al., 2005; Sedell et al., 1994). Riparian management, now a pillar of Pacific Northwest forest management (Young, 2000), is integral to balancing resource priorities and retaining biocomplexity at smaller scales (Lindenmayer and Franklin, 2002; Carey, 2006). The value of riparian management may be increased by piggy-backing the protection of terrestrial species upon aquatic–riparian protection strategies. At smaller spatial scales, a variety of species assemblages and ecological processes are relevant to consider, and likely require a mix of stream–riparian management approaches at the stand-to-drainage scale to tailor management to site and landscape conditions and landowner priorities. We review these species assemblages and ecological factors here, with an eye to synthesizing new information from Pacific Northwest headwater forests and their associated biota, focusing on riparian microclimates and amphibians, taxa uniquely suited for monitoring the health of stream and riparian environments (e.g., Welsh and Ollivier, 1998; Ashton et al., 2006). We also review current management approaches for headwater riparian zones in the U.S. Pacific Northwest, and address alternative management options for maintaining the persistence of headwater riparian-dependent amphibians, using a 2-tiered approach that provides: (1) a more protective design for target species in priority habitats within forested 6th-field watersheds; and (2) fewer species-protection measures where other resources such as timber are a priority.

The Pacific Northwest forested landscape extends continuously from northern California to Alaska, and in a more disjunct pattern from the Pacific Ocean across the interior Columbia River basin to the Rocky Mountains (Omernick et al., 2006). This enormous area, extending over 3,000,000 km^2, possesses diverse forest types and an equally diverse associated biota; stream–riparian zones across this range and their management approaches mirror this diversity. Naiman et al. (2000) regarded riparian forests of the Pacific Coastal Ecoregion as the most diverse vegetation assemblage in the region, supporting high wildlife diversity, including economically important fishes. These factors have been critical in focusing regional efforts to provide sound riparian management practices to sustain these systems, and have led to significant research to better understand riparian elements, particularly headwaters, the processes that structure them and the complex interactions among their biota.

New insights from science are redefining stream–riparian zones. Increasing knowledge of the lateral links among stream, riparian and upland biodiversity, and of the longitudinal interactions of biota from headwaters downstream is broadening the scope of our understanding of riparian ecosystem functions and processes (Wiens, 2002). Previously the lateral extent of riparian areas has been defined in terms of the inputs and processes important for fish (Gregory et al., 1991; Naiman and Décamps, 1997; Naiman et al., 2000). For example, distance from streams has been examined relative to contributions of down wood and litter, shading and water temperature retention, and bank stability or rooting strength of bank trees that is related to sedimentation (Fig. 1; from USDA and USDI, 1993; Naiman et al., 2000). Episodic events such as landslides contribute significant down wood to streams from far-upslope areas, whereas more chronic down wood inputs come primarily from within distances equal to 1/2 to 1 site-potential tree height (slope distance) away from streams (see Reeves, 2006).

This fish-centric perspective limits the focus of riparian habitats and their management to fish-bearing streams. However, recent studies have expanded the focus to explicitly consider the influence of upstream headwaters on downstream fish-bearing waters (Lowe and Likens, 2005). This is critical because fishless headwaters typically drain at least 70–80% of a catchment area (e.g., Gomi et al., 2002). Recent research has examined how headwater invertebrate production may provide a substantial prey base for downstream fishes (Wipfli and Gregovich, 2002), and headwater slope failures provide down wood (Reeves et al., 2003; Reeves, 2006) and sediment inputs (Benda and Cundy, 1990; Benda and Dunne, 1997a,b; Rashin et al., 2006) that fundamentally reinitiate streambed substrate "successional" sequences. Moreover, a diverse array of endemic amphibians inhabits headwaters in the Pacific

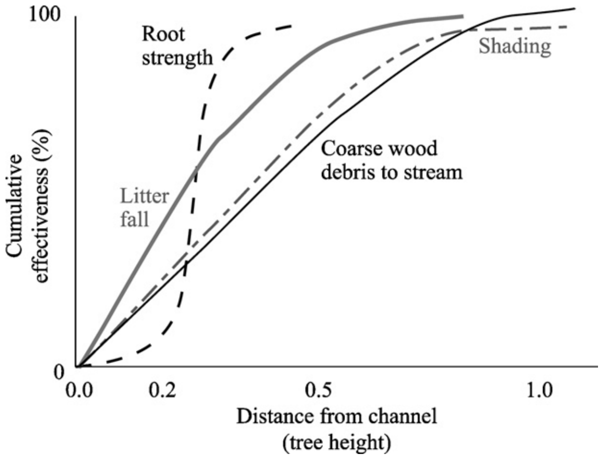

Fig. 1. Estimated distances upslope of streams of various components contributing to instream habitat conditions. Redrawn from USDA and USDI (1993).

Northwest, and only a nascent understanding exists of their role in stream–riparian systems. Increasing recognition exists that these amphibians have specific associations with instream, bank, and riparian microhabitats (Welsh and Lind, 2002; Olson and Weaver, 2007), and some species display habitat associations both longitudinally from ephemeral headwaters and zero-order basins (Sheridan and Olson, 2003) to fish-bearing reaches downstream (Hayes et al., 2006), and also laterally through the riparian zones to uplands (Rundio and Olson, 2007). Where they co-occur, these amphibians interact with fishes, potentially as part of cascading trophic networks (e.g., Rundio and Olson, 2003); laterally, similar interactions with terrestrial predators and prey are likely.

Stream amphibians fit well into the recently advancing perspective of ''reciprocal subsidies'' of stream–riparian zones (Davic and Welsh, 2004; Baxter et al., 2005), wherein stream biota are considered to provide ecologically important resources to streamside terrestrial areas, reciprocal to the terrestrially derived inputs into streams that were previously listed. For example, abundant aquatic diptera (flies) and plecoptera (stoneflies) emerge from the uppermost intermittent headwater channels (e.g., Progar and Moldenke, 2002; Muchow and Richardson, 2000) not only to drift downstream to fishes but to disperse into areas surrounding the riparian zone of headwaters. Welsh et al. (2005a) found higher abundances of coastal giant salamanders (*Dicamptodon tenebrosus*) and black salamanders (*Aneides flavipunctatus*) along intermittent headwater channels when contrasted with the downstream perennial reaches. Accessing invertebrate-rich intermittent headwater areas may also be one reason that coastal tailed frogs engage in upstream seasonal movements (Hayes et al., 2006). Similar situations may exist for post-metamorphic torrent salamanders (*Rhyacotriton*). Furthermore, diverse taxa such as carabid beetles along stream banks (Hering, 1998), bats along stream flyways (e.g., Swift et al., 1985), and birds (e.g., Gray, 1993) and snakes (Lind and Welsh, 1990, 1994) rely on aquatic prey. Aquatic nutrients are carried away from streams via these upland pathways. High densities and biomasses of stream amphibians imply that they play a significant role in stream–riparian dynamics (Bury, 1988). In streams with anadromous fishes, post-spawning, decaying fish carcasses provide ocean-derived organic matter inputs to uplands (e.g., Naiman et al., 2000). The subsidy these marine-accumulated materials provide to productivity of riparian areas is increasingly recognized. Hence, riparian areas are not simply the zones that directly influence fish habitat, but they constitute a highly concentrated nexus of dynamic and only partly recognized interactions among diverse aquatic and terrestrial biota with complex life histories.

1.1. Riparian habitats and microclimates

Scientific understanding of the discrete habitat conditions of headwater stream–riparian areas as well as the transition of conditions from wet stream to dry upslopes is rapidly evolving. Stream banks are recognized as sites of frequent disturbance resulting in relatively heterogeneous and complex microhabitat conditions. Microclimate differences contribute to the distinction of riparian environments from that of upland forest. However obvious this may seem, only recently has substantial research investment been made to characterizing riparian microclimate in the Pacific Northwest, particularly as it relates to ecological processes, habitat suitability, biodiversity and forest management in headwater forests. Literature on stream temperature is extensive (reviewed by Moore et al., 2005, and see below), but relatively less has been published on riparian microclimate. A few recent studies undertaken in the Pacific Northwest to characterize spatial and temporal variation in microclimate regimes of riparian headwater forests are beginning to provide information about the stream-channel-to-upslope continuum. Table 1 summarizes several of these studies with respect to air temperature and humidity.

Gradients in forest microclimate are common, particularly with respect to forest edges or topographic relief (Matlack, 1993; Chen et al., 1995). In riparian areas, open water surfaces, moist soils, and abundant vegetation contribute to the formation of microclimate gradients extending laterally from streams. Streams create a local environment through influences on air temperature and humidity. Streams directly influence air temperature by acting as either a thermal sink (day, warm season) or source (night, cool season). Near-surface water tables common to riparian areas indirectly influence microclimate by supporting development of vegetation and supplying moisture for transpiration from foliage.

In forest stands, summer daily maximum air temperature tends to increase, and daily minimum relative humidity tends to decrease with distance from headwater streams. These effects appear more pronounced in non-maritime locations (inland from the coast). Trans-riparian microclimate gradients are typically non-linear with greater rates of change near-stream and smaller rates of change with distance upslope. Several studies reveal that the strongest influence of the air temperature gradient is expressed within approximately 10–15 m upslope from the stream (Anderson et al., 2007; S. Chan et al., unpublished data [Oregon State University = OSU]; Rykken et al., 2007a; Welsh et al., 2005b). Generally the measured influence of streams on air temperature diminishes by distances of 30–60 m upslope of the stream in unharvested forests (Anderson et al., 2007; Brosofske et al., 1997; S. Chan et al., unpublished data [OSU]; Rykken et al., 2007a; Welsh et al., 2005b). Gradients in relative humidity generally show similar non-linearity, with a sharp near-stream gradient (Anderson et al., 2007; Brosofske et al., 1997; S. Chan et al., unpublished data [OSU]; Rykken et al., 2007a). However, Welsh et al. (2005b) described a more nearly linear trend of decreasing relative humidity with distance from the stream. In general, relative humidity gradients appear to extend further upslope than those of air temperature, but studies have rarely extended upslope microclimate monitoring far enough to make definitive comparisons.

1.2. Riparian biodiversity

Cool, moist conditions near streams provide habitat for many riparian-dependent species. Riparian plant assemblages

Table 1
Microclimate studies characterizing headwater stream riparian zones adjacent to various forest management practices in the Pacific Northwest and elsewhere

Microclimate variables measured	Measurement locations	Uncut buffer width (m)	Upslope condition	Stream width (m)	Region	Reference
Air temperature, relative humidity, soil/substrate, temperature	Stream center, ~6, ~15, ~25, ~50 m, and at 20 m intervals beyond 50 m up to a maximum of 230 m from stream center	6–70	Unthinned (~540 trees per hectare), thinned (99, 197, or 356 trees per hectare), or patch openings in 35–60-year-old Douglas-fir/western hemlock	0.3–3	Coast Range and western Cascade Range, Oregon, U.S.A.	Anderson et al. (2007)
Air temperature, relative humidity, surface air, temperature, soil temperature	Stream center, at buffer upslope edge, 15, 30, and 60 m upslope of the buffer edge, upland interior forest and in upland clearcut	0–72	Unharvested or clearcut Douglas-fir/western hemlock	2–4	Western Cascade Range, eastern Olympic Range, Washington, U.S.A.	Brosofske et al. (1997), Dong et al. (1998)
Air temperature, soil/substrate, temperature, relative humidity, soil moisture	Stream center and at 5, 15, 25, and 35 m from stream center		Clearcut, selectively thinned (100–200 residual trees per hectare), and intact second-growth Doulgas-fir		Coast Range, Oregon, U.S.A.	Chan et al. (unpublished data [OSU])
Relative humidity	Streamside and at 5, 10, 20, and 30 m from streamside	<30	Selectively harvested ponderosa pine, 5–51 m^2/ha residual basal area	1.3–5	East-side Cascade Range, Oregon and Washington, U.S.A.	Danehy and Kirpes (2000)
Air temperature	Lateral sampling at streamside, 10, 20, 30, 40, and 50 m from stream side; longitudinal sampling at 100 m intervals	25	Closed canopy mixed-wood forest, 80–100-year-old, or clearcut	First-order headwater	Western Maine, U.S.A.	Hagan and Whitman (2000)
Air temperature	Stream center, at centers of 5 and 30 m buffers, and in a clearing 20 m from buffer edge	5, 30	Closed canopy broadleaf forest or clearcut	4	Coromandel Peninsula, New Zealand	Meleason and Quinn (2004)
Air temperature, soil/substrate, temperature, relative humidity	1, 10, 20, and 70 m from stream center	30	Unthinned old-growth Douglas-fir/western hemlock, 5–22-year-old clearcuts with buffer and without buffer	1–5	West-side Cascade Range, Oregon, U.S.A.	Rykken et al. (2007a)
Air temperature, relative humidity	Streamside and at 10, 20, and 30 m from wetted stream channel	NA	Mixed grassland, second-growth Douglas-fir/hardwood forest and late-seral Douglas-fir/mixed hardwood forest	Intermittent and perennial streams	Mattole Watershed, eastern King Range, northern coastal California, U.S.A.	Welsh et al. (2005b)

also reflect topographic setting, position within the drainage network, disturbance history, local propagule sources, and the larger-scale ecological province. While analyses for other provinces in the Pacific Northwest region exist, many of the principal attributes of overstory and understory riparian plant assemblage structure at reach, basin and landscape scales are well-illustrated in the body of literature addressing forests of the Oregon Coast Range. Hence, we use many examples from this area to illustrate riparian plant biodiversity patterns.

The Oregon Coast Range ecoregion includes two major vegetation types, the western hemlock (*Tsuga heterophylla*) and the Sitka spruce (*Picea sitchensis*) associations (Franklin and Dyrness, 1988). In these forests, the abundance of shade-intolerant conifers such as Douglas-fir (*Pseudotsuga menziesii*) in the overstory tends to be lower near streams than upslope, over distances up to approximately 30 m from streams (Pabst and Spies, 1998, 1999; Minore and Weatherly, 1994; Nierenberg and Hibbs, 2000). Hardwood species, particularly red alder (*Alnus rubra*) are more likely to dominate overstory cover in near-stream areas of high fluvial activity, more recent disturbance, greater mineral soil exposure, and along wider streams (Minore and Weatherly, 1994; Pabst and Spies, 1999; Nierenberg and Hibbs, 2000). Near-stream sites initially occupied by red alder following disturbance frequently become dominated by salmonberry (*Rubus spectabilis*), a rhizomatous shrub and vigorous competitor, as the hardwood stands degenerate after 90–150 years (Nierenberg and Hibbs, 2000). These shrub-dominated riparian areas can be highly resistant to tree regeneration, but after some decades, shade-tolerant conifers may emerge through the shrub canopy.

Distinct understory communities develop for zero-order basins (i.e., unchanneled headwater swales or ‘‘headwalls’’), 1st- to 6th-order streams, and upslope forests (Pabst and Spies, 1998; Sheridan and Spies, 2005). Pabst and Spies (1998) defined 10 understory vegetation types from near-stream herb and shrub cover data across 1st- to 6th-order stream sites in the Coast Range of Oregon. They hypothesized that species composition was driven by valley-bottom to ridge-top environmental gradients and hillslope processes, particularly upslope gradients of decreasing soil moisture and relative humidity. They also identified three species groups that they considered strongly adapted to hydrological disturbance (periodic floods, alluvial deposition and erosion) as well as a separate group adapted to continuously saturated soils. Shade gradients also appeared to be a driver in the structuring of understory communities, including both stream-to-upslope gradients associated with increasing conifer canopy cover, and gradients along streams arising from a patchy distribution of canopy composition by conifers, hardwoods, shrubs and gaps. Sheridan and Spies (2005) found plant assemblages in zero-order basins tended to have a mix of species associated with both riparian areas and upslope forests. This likely reflects a spatial transition in hydrology associated with stream initiation and the establishment of a fluvial regime in addition to the topographic and shade gradient drivers that Pabst and Spies (1998, 1999) identified. Waters et al. (2001) reported a comparable pattern in a northern California watershed.

Diverse faunal groups also have strong stream–riparian associations. For example, Dunk et al. (2002) found mollusks occurred more frequently in riparian areas, and Rykken et al. (2007b) found a distinct invertebrate community occurring within 1 m of headwater streams. Naiman et al. (2000) noted that about 29% of wildlife species, including 34% of bird species in the Pacific Coast Ecoregion, require riparian areas; they are ‘‘riparian obligates.’’ In California, 40% of reptiles are riparian associates (Brode and Bury, 1984). Such habitat ‘‘associates’’ are typically defined as occurring in that habitat type more often or in higher abundances compared to alternative habitats. Riparian associates may or may not be riparian obligates, but appear to have some dependency on riparian habitats. Kauffman et al. (2001) reported 319 of 593 (53%) of general wildlife in Oregon and Washington were riparian associates. Thomas et al. (1979) found the proportion was higher for vertebrates in the Oregon Blue Mountains (285 of 378 [75%] species) and the Oregon Great Basin (287 of 363 [79%]). In the arid U.S., riparian areas may account for less than 1% of the landscape, but in northern Colorado, riparian areas support 82% of breeding birds (Knopf and Samson, 1994). These numbers include species occurring around both lentic and lotic waters, but it highlights the nature of riparian zones as an interface of biodiversity, where upland terrestrial forms use the aquatic–riparian zone incidentally or for some life history functions, and where riparian- and aquatic-dependent forms reside.

For amphibians, this scenario is echoed and amplified. While a third of the approximate 265 amphibian species north of Mexico are stream-associated (Corn et al., 2003), all 47 Pacific Northwest species are either obligate or facultative stream–riparian associates (Table 2; Jones et al., 2005; Lannoo, 2005). Fifteen (32%) of these species are obligate forms tied to stream or riparian areas for life history functions, whereas the remaining 68% are facultative species that display more incidental use (e.g., if suitable habitat conditions exist along stream for their life history functions). Twelve of the 15 obligate forms occur primarily in headwater streams. While the relative use of headwaters versus larger streams is documented for headwater-associated species, the reasons for this association is not well known; however, these animals often have strong habitat associations with elements of headwaters (e.g., steep gradients, coarse substrates) and are proposed to have predation-structured assemblages (e.g., downstream predators such as fishes and giant salamanders may displace prey into headwaters).

This suite of 47 species includes stream- and pond-breeding amphibians and bank dwellers. These species often use riparian zones upon metamorphosis for foraging or dispersal. Salamander species that breed in upslope terrestrial habitats may only coincidentally occur along streams if their habitat transects these areas. Some terrestrial-breeding species shift from a facultative to a more obligate use of riparian zones as upland habitat and climate change across their range. For example, the black salamander, *A. flavipunctatus*, was highly riparian-associated relative to uplands at interior sites in northern California (Nauman and Olson, 2004) compared to

Table 2
Pacific Northwest amphibians occurring in streams and riparian zones

Species	Stream/riparian association	Comments
Frogs and toads		
Ascaphus montanus Rocky Mountain tailed frog, ASMO	Obligate stream breeding, rearing; riparian foraging	Moderate-to-fast flow headwater streams in forests; often step-pools, cold water, coarse substrate
Ascaphus truei Coastal tailed frog, ASTR	Obligate stream breeding, rearing; riparian foraging	Moderate-flow headwater streams in forests; often step-pools, cold water
Bufo boreas Western toad, BUBO	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater breeding in forests, grasslands, meadows, deserts
Bufo woodhousii Woodhouse's toad, BUWO	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater habitat breeding in grassland, shrub steppe
Pseudacris maculata Boreal chorus frog, PSMA	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater habitat breeding in meadows, swamps and woodlands
Pseudacris regilla Pacific treefrog, PSRE	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater habitat breeding in forests, woodlands, grasslands and other areas; shorter hydroperiod
Rana aurora Northern red-legged frog, RAAU	Stream breeding, rearing if habitat available; riparian foraging and migration	Diverse stillwater habitat breeding in forests; longer hydroperiod
Rana boylii Foothill yellow-legged frog, RABO	Obligate stream breeding, rearing, foraging, and overwintering; riparian foraging	Typically 3rd order-streams or larger for breeding; also uses smaller tributaries; coarse substrates; in forested foothills
Rana cascadae Cascades frog, RACAS	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater habitat breeding in montane forests; longer hydroperiod
Rana catesbeiana Bullfrog, RACAT	Stream breeding if habitat available; riparian foraging	Permanent, warm, stillwater habitat used for breeding; non-native
Rana clamitans Green frog, RACL	Stream breeding if habitat available; riparian foraging	Diverse stillwater habitat for breeding; non-native
Rana draytonii California red-legged frog, RADR	Stream breeding, rearing, foraging if habitat available; riparian foraging	Diverse stillwater habitat breeding in forests, grasslands, agricultural lands; often with emergent vegetation
Rana luteiventris Columbia spotted frog, RALU	Stream breeding, rearing if habitat available; limited riparian foraging	Diverse stillwater habitat breeding in forests, grasslands and brushlands; longer hydroperiod
Rana pipiens Leopard frog, RAPI	Stream breeding, rearing if habitat available; riparian foraging and migration	Diverse stillwater habitat breeding in grasslands, shrublands or meadows in forests
Rana pretiosa Oregon spotted frog, RAPR	Stream breeding, rearing if habitat available; riparian foraging and migration	Low-emergent marsh stillwater breeding in grasslands and forests
Rana sylvatica Wood frog, RASY	Stream breeding, rearing if habitat available; riparian foraging and migration	Diverse stillwater habitats used for breeding in forests
Spea intermontana Great Basin spadefoot, SPIN	Stream breeding if habitat available; riparian foraging	Diverse stillwater breeding habitat in grassland, shrub steppe, woodlands and forests
Woodland salamanders		
Aneides ferreus Clouded salamander, ANFE	Facultative riparian breeding and foraging	Large decayed wood in or near forests probable breeding requirement, probable plasticity in habitat needs, associated with headwalls of zero-order basins
Aneides flavipunctatus Black salamander, ANFL	Facultative riparian breeding and foraging	Probable wood-linked or talus breeding site in forests, riparian- and headwater-associate at interior sites
Aneides lugubris Arboreal salamander, ANLU	Facultative riparian breeding and foraging	Wood-associate, to 30 m above ground in oak woodlands and forests
Aneides vagrans Wandering salamander, ANVA	Facultative riparian breeding and foraging	Wood associate, to 90 m above ground in forests
Batrachoseps attenuatus California slender salamander, BAAT	Facultative riparian breeding and foraging	Probable wood-linked breeding site in forests, grasslands, chapparal
Batrachoseps wrighti Oregon slender salamander, BAWR	Facultative riparian breeding and foraging	Wood associate in forests, also in talus
Ensatina eschscholtzii Ensatina, ENES	Facultative riparian breeding and foraging	Wood and talus associate in forests
Hydromantes shastae Shasta salamander, HYSH	Facultative riparian breeding and foraging	Often limestone-associated in forests, also other rock, down wood
Plethodon asupak Scott Bar salamander, PLAS	Facultative riparian breeding and foraging	Rock-associated in forests
Plethodon dunni Dunn's salamander, PLDU	Obligate riparian breeding, rearing, and foraging	Often rock-associated in forests; may occur in upland forest
Plethodon elongates Del Norte salamander, PLEL	Facultative riparian breeding and foraging	Rock-associated in forests

Plethodon idahoensis Coeur d'Alene salamander, PLID	Obligate riparian breeding, rearing, and foraging	Often rock-associated in forests
Plethodon larselli Larch Mountain salamander, PLLA	Facultative riparian breeding and foraging	Often rock-associated in forests
Plethodon stormi Siskiyou Mountains salamander, PLST	Facultative riparian breeding and foraging	Rock-associated in forests
Plethodon vandykei Van Dyke's salamander, PLVA	Obligate riparian breeding, rearing, and foraging	Rock and wood associated in forests
Plethodon vehiculum Western redback salamander, PLVE	Facultative riparian breeding and foraging	Rock and wood associated in forests, along stream banks
Other salamanders		
Ambystoma gracile Northwestern salamander, AMGR	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater habitat breeding in forests, grasslands and agricultural lands; longer hydroperiod
Ambystoma macrodactylum Long-toed salamander, AMMA	Stream breeding, rearing if habitat available; V	Diverse stillwater habitat breeding in forests, sagebrush and alpine meadows; shorter hydroperiod
Ambystoma tigrinum Tiger salamander, AMTI	Riparian foraging and migration	Diverse stillwater habitat breeding in grasslands, savannahs and woodland edges
Dicamptodon aterrimus Idaho giant salamander, DIAT	Obligate stream breeding, rearing, foraging; riparian foraging	Moderate-to-fast flow headwater streams in forests; often step-pools, cold water, some lakes
Dicamptodon copei Cope's giant salamander, DICO	Obligate stream breeding, rearing, foraging; riparian foraging	Moderate-flow headwater streams in forests; often in pools, cold water
Dicamptodon ensatus California giant salamander, DIEN	Obligate stream breeding, rearing, foraging; riparian foraging	Moderate-flow headwater streams in forests; often step-pools, cold water
Dicamptodon tenebrosus Coastal giant salamander, DITE	Obligate stream breeding, rearing, foraging; riparian foraging	Moderate-to-low flow streams in forests; often small streams, in pools, cold water
Rhyacotriton cascadae Cascade torrent salamander, RHCA	Obligate stream breeding, rearing, foraging; riparian foraging	Headwater stream focused in forests; low-flow habitats, cold water
Rhyacotriton kezeri Columbia torrent salamander, RHKE	Obligate stream breeding, rearing, foraging; riparian foraging	Headwater stream focused in forests; low-flow habitats, cold water
Rhyacotriton olympicus Olympic torrent salamander, RHOL	Obligate stream breeding, rearing, foraging; riparian foraging	Headwater stream focused in forests; low-flow habitats, steep slopes, cold water
Rhyacotriton variegates Southern torrent salamander, RHVA	Obligate stream breeding, rearing, foraging; riparian foraging	Headwater stream focused in forests; low-flow habitats, cold water
Taricha granulose Roughskin newt, TAGR	Stream breeding, rearing if habitat available; facultative riparian breeding, foraging and migration	Diverse stillwater habitat breeding in or near forests; longer hydroperiod
Taricha rivularis Red-bellied newt, TARI	Obligate stream breeding, rearing, foraging; riparian foraging and migration	Moderate-to-fast flow streams in redwood forests and oak woodlands; rocky substrates, cold water
Taricha torosa California newt, TATO	Stream breeding, rearing if habitat available; riparian foraging	Diverse stillwater breeding in grasslands, chaparral, forests and oak woodlands

Species acronym represents first letters of genus and species names. Data from Jones et al. (2005) and Lannoo (2005).

coastal areas where it does not always appear to be a riparian associate (except see Welsh et al., 2005a). The relative use of riparian versus upland habitats across species ranges is poorly explored for many amphibians with life history stages that occupy terrestrial habitats.

In the Pacific Northwest, most (89% [$n = 42$]) amphibian species occur in forests. Hence riparian areas in Pacific Northwest forests are indispensable to this taxonomic group in the region. Furthermore, salamanders have been proposed to be ecologically significant due to their: (1) presumed sensitivity to historic forest practices (e.g., deMaynadier and Hunter, 1995; Ashton et al., 2006); (2) central location within food webs because they serve as both dominant predators and prey; (3) high densities; and (4) sensitivity to microclimate and microhabitat conditions (e.g., Bury, 1988; Bury and Corn, 1991). They are proposed to be used as metrics of forest ecosystem health (Davic and Welsh, 2004; Welsh and Droege, 2001), with stream amphibians as specific indicators of aquatic ecosystem stress (Welsh and Ollivier, 1998). Global declines of amphibians were recognized over a decade ago (Blaustein and Wake, 1990), with an updated global assessment (Stuart et al., 2004) reviving concerns worldwide. In Pacific Northwest forests, stream–riparian breeding amphibians may be particularly indicative of stressors intersecting aquatic and terrestrial systems (e.g., Welsh et al., 2005a). Such trends, in tandem with new scientific information about ecological associations and mechanisms in headwater streams, indicate that close examination is warranted of the effectiveness of riparian management in preserving these animals and the ecosystems they inhabit.

1.3. Use of riparian and upland habitats

Integrating stream, riparian and upland habitat use by amphibians for the delineation of riparian boundaries is not straightforward. Riparian areas may function as habitat for resident species and as corridors for transient species (e.g., Rosenberg et al., 1997). Discrete habitats for breeding, foraging and overwintering are now documented for Pacific Northwest amphibians such as the Columbia spotted frog (Pilliod et al., 2002), which can use select streams for breeding but is predominantly a pond-breeder. This reliance on multiple, potentially disparate locations within a local area for all life history functions is well known for terrestrial vertebrates (e.g., Law and Dickman, 1998) and warrants further study in stream- and riparian-associated species.

The lateral spatial extent of habitats used by aquatic- and riparian-dependent organisms is becoming better understood, although the distance to which aquatic organisms (i.e., ‘‘subsidies’’ in the reciprocal subsidy model) reach upslope or from which terrestrial organisms reach the stream (e.g., Nakano and Murakami, 2001) are not well known (Polis et al., 2004). Several studies in the PNW have captured northwestern stream-breeding or riparian-dependent amphibians away from streams (Table 3), suggesting that regular movement overland

Table 3
Maximum distances away streams into upland forests at which Pacific Northwest amphibians have been found

Species	Distance (m)	Comments	Reference
RHKE/RHVA, ANFE, PLDU, PLVE, ENES, TAGR [ASTR, DITE]	30–40 m [20–30 m]	Time-constrained searches in quadrats 0–10, 10–20, 20–30, and 30–40 m from streams; 5–100+ year old conifer stands, 0 to 40+ m buffers, 1st–3rd order streams, Oregon; DITE, RHKE/RHVA, PLDU and ASTR found primarily at 0–10 m from stream, PLVE found primarily at 0–20 m from stream	Vesely (1996)
AMGR, TAGR, ENES, PLVE	55	Pitfall traps and cover boards spanning 5, 30 and 55 m from stream; 70-year-old conifer stands; British Columbia; more AMGR were captured at 30 m from stream than 5 and 55 m from stream; AMGR and TAGR moved more along streams than up or down slopes	Maxcy and Richardson (2000)
ASTR	65	Pitfall traps at 5, 25, 45, 65 m from stream; <5 and 80+ year old conifer stands; non-fish-bearing streams <3 m wide; British Columbia; no difference in capture frequency across distance from stream	Matsuda and Richardson (2005)
DITE	66, 22, 19	Radiotracking; forested, clearcut with buffer (20–30 m), and clearcut conifer stands; small strems 1–6 m wide; British Columbia; maximum distance from stream was 66 m in intact forested habitat, 22 m in clearcut, 19 m in buffered clearcut	Johnston and Frid (2002)
ASTR	100	Pitfall traps 5, 25, 50 and100 m from stream; 5- and 250-year-old conifer stands; headwater streams 1–3 m wide and fishless; British Columbia; mean distances from stream reported (adults: 28 m; juveniles: 17 m; males 23 m; females 17 m); frogs captured farther from streams in old growth than in clearcuts	Wahbe et al. (2004)
DITE, TAGR, PLVE, ENES, AMGR, ASTR, RAAU	∼135	Pitfall traps, 10 × 10 grid across 135 m × 135 m; 40–50-year-old red alder stands; 2nd order streams, Oregon	McComb et al. (1993a)
ASTR, DITE, BAWR, PLDU, PLVE, RAAU, RHsp, TAGR	200	Pitfall traps at stream and 200 m upslope, 2 trans-riparian pitfall arrays with traps every 25 m; 5–200+ year old conifer stands, deciduous stands; 3rd–4th order streams, Oregon	Gomez and Anthony (1996)
DITE, TAGR, PLDU, PLVE, ENES, ANFE, ASTR, RAAU	400	Pitfall traps <10 and 400 m from streams, 2 trans-riparian arrays with traps every 50 m; 12–140 year conifer stands; 2nd–3rd order streams, Oregon	McComb et al. (1993b)

Species acronyms follow Table 2.

may occur. Several stream obligate species have been detected 100 m or more from streams (e.g., 100 m, Wahbe et al., 2004; 135 m, McComb et al., 1993a; 200 m, Gomez and Anthony, 1996; 400 m, McComb et al., 1993b). Occurrence of stream–riparian obligates 200–400 m away from streams implies they could cross ridgelines where these come in close proximity to stream channels. Wahbe et al. (2004) found geographic variation in upland movements of coastal tailed frogs, *Ascaphus truei*, in that they remained close to streams at inland sites (e.g., 12 m) but some individuals moved farther from streams (>100 m) at coastal sites. This also may be the case with the black salamander mentioned previously, which has riparian associations inland (Nauman and Olson, 2004; Bury, 2005) but is not generally considered a strict riparian associate. More mesic coastal climate conditions may ameliorate surface microclimates, fostering movements. Wahbe et al. (2004) found that forest condition influenced tailed frogs with more animals occurring along streams and fewer in upland sites in clearcuts in comparison to older forests. Johnston and Frid (2002) also found coastal giant salamanders, *D. tenebrosus*, closer to streams in clearcuts than in forested stands. Western redback salamanders were more restricted to riparian zones in second growth compared to old-growth forests (Dupuis, 1997). These patterns may indicate greater site fidelity to water sources and decreased dispersal tendencies for these animals on lands managed for timber (Johnston and Frid, 2002). Alternatively, the results could reflect altered timing of movements. Movements may have been restricted by microclimate in the generally warmer and drier clearcuts, and to rainy times of dry years (Dupuis et al., 1995; Johnston and Frid, 2002). Understanding these altered life history or behavioral attributes may affect population structure and dynamics, and be important to consider for the long-term persistence of animals across managed landscapes.

Some degree of mobility also is known for other northwestern amphibians, but mobility may be coupled with site specificity at other life stages or seasons. The red-bellied newt, *Taricha rivularis*, a stream breeder at the southern extent of our area of consideration, appears to have high site fidelity and also high mobility, returning to natal streams for breeding even if displaced several kilometers (Kuchta, 2005). Maxcy and Richardson (2000) found the closely related roughskin newt, *Taricha granulosa*, and the northwestern salamander, *Ambystoma gracile* traveled more often along streams than up or down slopes. This may result from directional movement toward or away from their lentic breeding sites (Maxcy and Richardson, 2000), suitable conditions for surface movements along streams (e.g., cool, moist conditions), or if streams serve as full or partial barriers to movement, diverting travel along them and hence resulting in higher animal abundances. A seasonal upstream movement has been documented for the coastal tailed frog that likely represents a seasonal cycling between breeding habitat in the perennial portion of the headwater stream and foraging habitat near the upstream origin (Hayes et al., 2006). Thus, while spatial extent of a "population" is still a mystery for most stream–riparian associated amphibians in this region, evidence of their use of uplands and movements along riparian corridors is accumulating. Both longitudinal and lateral connectivity of habitats is likely important for long-term persistence of aquatic–riparian species and assemblages. Wahbe et al. (2004) suggested that a tailed frog population may extend across multiple streams in a watershed, a suggestion that agrees with the recent finding that genetic connectivity in tailed frogs can extend up to 20–30 km in continuous habitat (S. Spear and A. Storfer, personal communication). Johnston and Frid (2002) estimated the home range of a single coastal giant salamander was 935 m^2 in intact forests, so that a population could cover a much more extensive area, and potentially cover multiple streams. Terrestrial-breeding salamanders appear less mobile. For example, mark-recapture studies have reported that ensatina (*Ensatina eschscholtzii*), the Del Norte salamander (*Plethodon elongatus*), California slender salamander (*Batrachoseps attenuatus*), and the western redback salamander appear to move over spatial scales <10 m (Hendrickson, 1954; Ovaska, 1988; Ovaska and Davis, 2005; Karraker and Welsh, 2006). At these smaller spatial scales, riparian forest may serve as important habitat for these species. However, these species may be abundant upslope, and loss or degradation of upland forest habitat could disrupt population dynamics or affect persistence.

Conceptually, disrupted habitat connectivity, or habitat fragmentation, can affect populations through either within-fragment processes (e.g., demographic, genetic, and stochastic effects) or between-fragment processes (e.g., source–sink or metapopulation dynamics; Davies et al., 2001; Pullin, 2002). Several studies have demonstrated fragmentation effects on amphibians (Cushman, 2006), although the research on these topics is sparse for amphibians in contrast to other animal groups (Lidicker and Koenig, 1996; McGarigal and Cushman, 2002). Moreover, most information on habitat and population isolation, metapopulation dynamics, and dispersal or movement among amphibians comes primarily from pond-breeding rather than stream- or terrestrial-breeding species (e.g., Smith and Green, 2005; Cushman, 2006). Many studies reveal pronounced decreases in the ability of amphibians to disperse as human alteration of landscapes increases, and survival of juveniles in such fragmented landscapes is often substantially reduced (see Cushman, 2006). These findings imply dramatic effects on immigration because dispersal in amphibians is thought to be primarily effected by juveniles rather than adults (e.g., Funk et al., 2005; Cushman, 2006). For Pacific Northwest forest amphibians occurring in headwaters, there is some indirect support that reduced population connectivity might be associated with timber harvest; for example, the previously described patterns of restricted distribution to streams of tailed frogs and coastal giant salamanders in clearcut areas might reduce connectivity at larger scales (Wahbe et al., 2004; Johnston and Frid, 2002). However, the relative effects on population persistence of site-specific habitat alteration for the various life history stages of headwater species and altered between-site connectivity are unknown.

Finally, amphibians can have a metapopulation dynamic in which local populations occasionally become extirpated but the site is eventually recolonized by individuals from nearby populations (Alford and Richards, 1999; Sjögren Gulve, 1994).

In theory, a stable metapopulation relies on longer-term interactions among such sites. Although the generality of this concept has been challenged for some amphibians (Smith and Green, 2005), ample evidence exists of a meta-structure in many amphibian populations. A metapopulation structure may place a population at greater risk (Heinz et al., 2006) because it relies on the long-term dynamics among organisms at many localities in an area. Hence, it may take a more thoughtful or complex forest management strategy to maintain the population. In particular, in forested landscapes, the extent, pattern and duration of disrupted connectivity are important considerations. For example, timber harvest may be a disturbance with a relatively short duration because many forest conditions (e.g., canopy cover, microclimate regimes) become re-established within years-to-decades as trees regenerate. The effects of episodic disturbances are not well known for most species.

A few studies address these concepts with headwater stream-breeding amphibians, both in the Pacific Northwest and in the eastern U.S. For example, in North Carolina, Willson and Dorcas (2003) found the relative abundance of headwater salamanders was inversely proportional to the percentage of disturbed habitat in the sub-watershed of the stream. They concluded that protection of headwater stream populations should consider watershed-scale land use. In New Hampshire, Lowe and Bolger (2002) found the abundance of spring salamanders in isolated headwater streams was less than in paired streams having a downstream tributary junction linking the streams. They suggested that landscape connectivity confers resilience to habitat disturbance. In the U.S. Pacific Northwest, both Stoddard and Hayes (2005) and Raphael et al. (2002) detected larger spatial scale effects on stream animal abundances. At drainage scales, Stoddard and Hayes (2005) found coastal tailed frog and coastal giant salamander abundances were associated with the proportion of stream lengths having forested bands >46 m wide. Torrent salamanders lacked a drainage-scale association, but were associated with these stream buffer widths at a smaller spatial scale. Welsh et al. (2005a,b) linked changes in abundances and species composition of stream-associated amphibian assemblages in the Mattole watershed of northwestern California to changes in stream temperature regimes resulting from landscape-level fragmentation due to the harvesting of late-seral forests, and the conversion of forests to small homesteads and rangeland. Most recently, identification of whether coastal tailed frogs use headwater basins for reproduction in southwestern Washington appears to be related to the extent of non-fish-bearing stream habitat in particular basins (Hayes et al., 2006).

Fragmentation of Pacific Northwest forests from timber harvest practices can be expected to affect stream–riparian amphibians with both long- and short-distance dispersing capabilities (Heinz et al., 2006). Cushman (2006) concluded from a number of studies that the short-term impact of habitat loss and fragmentation is greater for long-distance than short-distance dispersing amphibians, apparently because of high mortality rates for dispersing individuals in the matrix of altered habitat. However, short-distance dispersers have greater likelihood for isolation than long-distance dispersers, which greatly increases their risk of local extirpation. Too few data exist to definitively evaluate the relative risk of general long- versus short-distance dispersal categories. Frequent extirpation and turnover may represent a typical background phenomenon among populations of a number of amphibian species, indicating that ‘‘population connectivity is ultimately important even for populations of species that are not directly impacted by habitat loss or elevated mortality risks in dispersing’’ (Cushman, 2006).

Genetic studies are increasingly used to frame conservation decisions (Hedrick, 2001; Frankham, 2003), and can provide insights about the scale of historical connectivity among populations. Molecular genetics represents a tool that provides critical population data that are, at best, difficult to obtain with even the most sophisticated demographic methods. Recently, genetic data were used to evaluate the population structure of coastal giant salamanders in managed landscapes in British Columbia (Curtis and Taylor, 2003). That analysis identified decreased genetic variability with an increase in area over which forestry practices had been applied. While demographic data are used to assess the immediate health of a population, only genetic data can identify genetic variability. Reduced genetic variability (e.g., from reductions in population size or inbreeding) is thought to have negative consequences for population function and ultimately, species survival (Schrader-Frechette and McCoy, 1993; Frankham, 2003). Hence, estimating genetic diversity, in addition to demographic population size estimates, gives a more rigorous basis for making predictions about short- and long-term survival of species in response to land use changes. However, a current challenge is to understand how genetic diversity, or specifically a reduction in genetic variation, might predictably affect population stability; in particular, a risk analysis for loss of genetic diversity for headwater amphibians has not been conducted.

Ultimately, populations are defined both demographically and genetically, but genetic data can most-precisely identify the scales over which gene flow occurs among populations (Frankham, 2003). Empirically defined ‘‘genetic neighborhoods’’ or inferred spatial patterns of gene flow are basic yardsticks for not only identifying the spatial footprint of populations, but also for providing insights into how landscape features or land management practices may constrain populations. Genetic population structure data are currently unavailable for most amphibian species in the Pacific Northwest, but a few data are beginning to emerge. For example, the spatial scale of the genetic neighborhood of the stillwater-breeding Cascades frog (*Rana cascadae*), sampled across a broad range of populations, was about 10 km (Monsen and Blouin, 2004). As previously mentioned, a 20–30 km estimate was recently obtained from preliminary data on coastal tailed frogs (S. Spear and A. Storfer, personal communication). Such genetic neighborhood data take on particular importance because they define a spatial scale at which management should be addressed (Frankham, 2003).

Genetics also can contribute two other important pieces of data on population structure unobtainable by other means. First, they shed light on whether local populations are autonomous or represent elements linked in some kind of meta-structure (Smith and Green, 2005), and second, for those populations that display some kind of meta-structure, they can indicate which of the member populations represent recent ‘‘sources’’ or ‘‘sinks’’ in that structure (Frankham, 2003). With the appropriate reference populations, basic data on genetic structure allows identification of distinctive genetic patterns that provide clues to genetic health (e.g., inbreeding levels) or potential legacy effects that may have either anthropogenic or non-anthropogenic origins. Assuming the observed dynamics are somewhat stable over time, the ability to identify the populations that donate migrants (‘‘sources’’) would aid management decisions to maintain viable meta-populations. Currently, genetic analyses identifying meta-populations and their source–sink relationships have not been conducted for Pacific Northwest amphibians. Until such studies are available for stream-associated amphibians or other taxa, our delineation of ‘‘riparian’’ based on biotic use patterns will need to rely on inference from less powerful observations, such as captures away from streams (e.g., Table 3). These types of data could have a large impact on the landscape designs; managers could more efficiently target the best areas for protection, and yield greater success in species management for long-term persistence. This approach might be considered for those headwater taxa of greatest concern.

2. Management approaches

2.1. Current management systems

Current management approaches for forested stream–riparian zones in the Pacific Northwest constitute a variable mix of land management jurisdictions and natural resource priorities (Fig. 2; Table 4). Key elements of current approaches include delineation of management zones in which timber harvest is not allowed (no-cut, Fig. 2) or where harvest is allowed with limitations of equipment use or levels of tree removal (management zone, Fig. 2; Table 4; Young, 2000). Rationale for riparian management zone delineations can typically be traced back to recommendations for: (1) retaining stream bank stability (~10 m) to reduce sedimentation; (2) maintaining instream habitat attributes such as water temperature, litter and wood inputs (~15–30 m); and (3) a more conservative approach for provision of instream habitat conditions with benefits to riparian-dependent species (~40–100 m). In many regards, these measures have not been well tested, and hence represent application of our best available science relative to diverse stream–riparian priorities. Implementation of diverse measures may represent opportunities for monitoring their effects. For example, one recent study (Rashin et al., 2006) examined Washington State management practices and found that 94% of erosion factors associated with sediment delivery to headwater streams were located within 10 m of streams, supporting the value of a near-stream buffer to reduce sedimentation impacts.

Comparing riparian management rules among different plans and jurisdictions (Fig. 2; Table 4) is not simple because management rules vary among multiple spatial and practical dimensions. Perhaps the foremost consideration, however, is categorical: most policies have different rules for streams with and without fish, and with perennial or ephemeral flows. The second major dimension of interest is the width of the overall zone that is targeted for some kind of ‘‘special’’ management. The simplest way to characterize this zone is by reference to its width relative to stream axis or the water’s edge. With this information coupled to mapping of waterways for a given locale, it is possible to map and estimate the overall area managed for riparian-specific purposes (e.g., Fig. 2). The third critical dimension can be summarized as the set of practices allowed within the designated area, including the guiding objectives that determine in particular cases whether specific practices are allowed or prohibited. The importance of overarching objectives is exemplified in the Northwest Forest Plan Aquatic Conservation Strategy (USDA and USDI, 1994). Because the dictum of management within the designated riparian area is of ecological benefit to habitat and water quality values, and because any management within this area is complicated by both diverse and frequent natural disturbance processes and severely constrained by pervasive past human alteration that greatly depleted large woody debris, ‘‘active management’’ such as timber harvest needs careful consideration, and in some cases may be difficult to justify.

The various approaches to riparian forest management in headwater areas (Fig. 2; Table 4) reflect legal mandates and political influences that vary according to land ownership and have been in flux in recent years. On U.S. National Forests, for example, the National Forest Management Act of 1976 imposed a mandate to protect biological diversity, maintain or improve water quality, and in particular, prevent harmful delivery of sediment to streams. Although the U.S.D.A. Forest Service has interpreted the biological diversity requirement as the need to maintain viable populations or a reasonably secure regional distribution of native species, the implications of the policy with respect to headwater stream biota, including amphibians, remains poorly understood. In 2005, a new planning rule was instituted for the National Forest System (US 36 CFR Part 219; Federal Register, 2005) that addresses biological diversity protections using ecosystem approaches first, with additional provisions for threatened or endangered species, species-of-concern and species-of-interest. Under this new rule, some headwater stream species are likely candidates to be considered species-of-concern or interest, but species in this assemblage are only recently gaining recognition as themselves constituting natural values that warrant recognition in forest management.

Federal, state and private forestry initiatives have long focused on fish because many stream-rearing fish are highly valued for recreational and commercial purposes. As a result, a considerable body of science has developed linking specific categories of forest practices to impacts on fish habitat. However, the logic of focusing only on fish-bearing reaches for designing management prescriptions to protect fish has been challenged (Welsh, 2000) on the grounds that instream fish habitat quality is more influenced by upstream conditions and processes in the non-fish-bearing reaches than by those at the

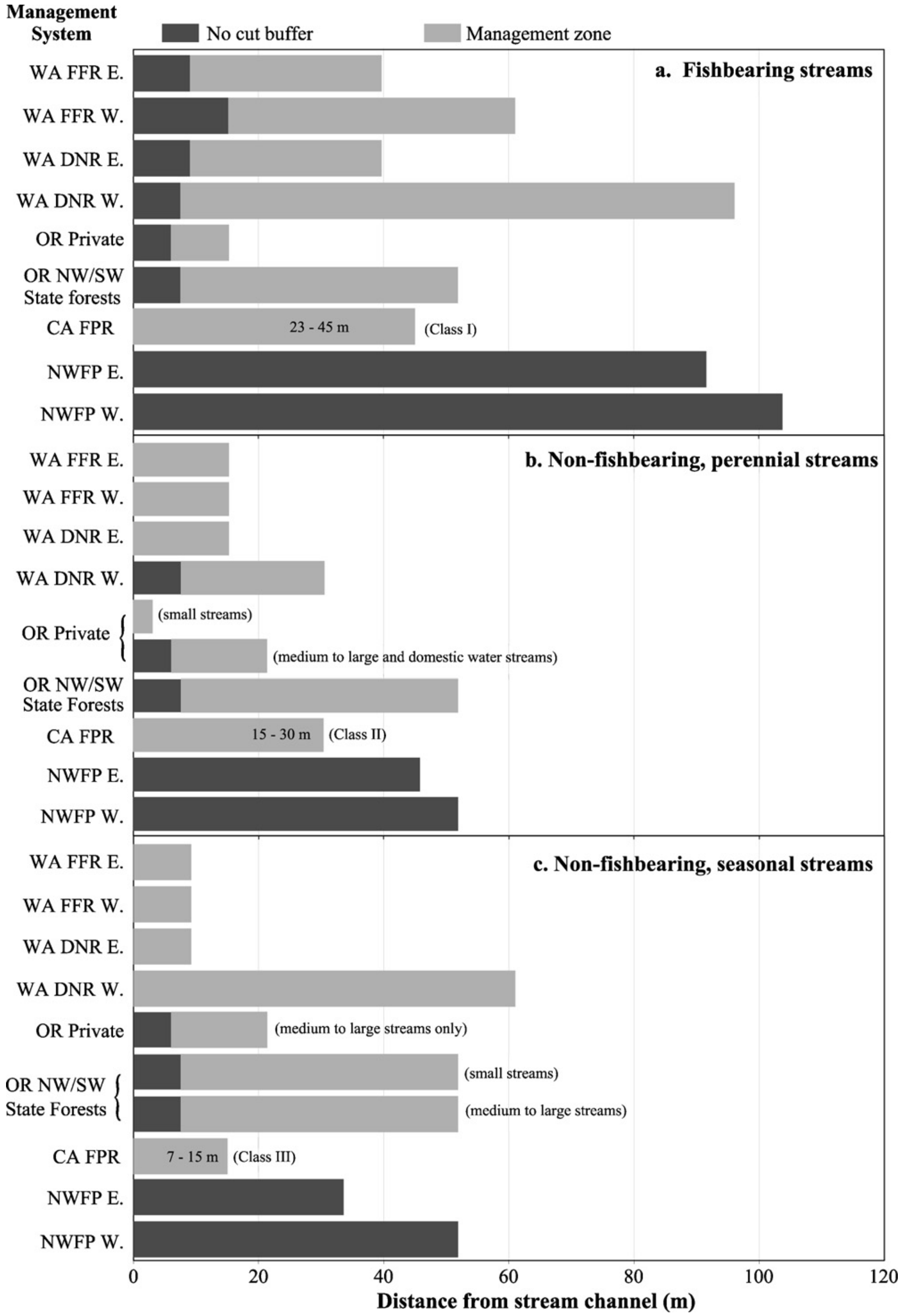

Fig. 2. Management systems in the U.S. Pacific Northwest delineating riparian forest management zones. Table 4 provides additional information for each system.

streamside of fish-bearing reaches (see Montgomery, 1999). Additionally, fish distributions may have been altered by human disturbances, such that fish-bearing reaches identified today may be a limited view of the historic condition, with some species suffering diminished distribution in headwater streams. For example, only recently has biological passage through road-crossing culverts become a concern. On U.S. Forest Service and Bureau of Land Management roads in Oregon and Washington, over half of ∼10,000 culverts on fish-bearing streams were determined to be barriers to salmonids (US GAO, 2001); hence some fishless headwater reaches may result from artificial barriers of downstream road crossings. Welsh et al. (2000) concluded that sedimentation from unprotected upstream reaches pushed fish distributions to downstream reaches in California. Others, such as Jackson et al. (2001) and Rashin et al. (2006) in Washington, have reported that sedimentation of unbuffered headwater streams altered stream habitats. They considered sedimentation in headwater streams to have particularly adverse consequences for stream amphibians.

Table 4
Riparian buffer widths delineated by various management systems in U.S. Pacific Northwest forests

Management system	No cut buffer (m)	Management zone (m) with timber harvest allowed	Comments	References
Perennial fish-bearing streams				
Washington Forest and Fish Report, West of Cascades WA FFR W	15	3–30: inner zone; 9–20: outer zone		Washington Administrative Code (2006) [WAC 222-30-021(1)]
Washington Forest and Fish Report, East of Cascades WA FFR E	9	14–21: inner zone; 0–17: outer zone		Washington Administrative Code (2006) [WAC 222-30-022(1)]
Washington Department Natural Resources, West of Cascades WA DNR W	8	23: inner zone; 0–66: outer zone		Washington State Department of Natural Resources (1997)
Washington Dept. Natural Resources, East of Cascades WA DNR E	9	14–21: inner zone; 0–17: outer zone		Washington Administrative Code (2006) [WAC 222-30-022(1)]
Oregon Private lands	6	9–24		Oregon Administrative Rules (2006) [OAR 629-640-100 (2)(b); OAR 629-635-0310 (1)(a)]
Oregon NW/SW State lands	8	23: inner zone; 21: outer zone		Oregon Department of Forestry (2001)
California Forest Practice Rules CA FPR	None	23–45	Class I streams. Varies with steepness of side-slope; 50% minimum canopy cover for both overstory and understory	Young (2000)
Federal Northwest Forest Plan, East of Cascades NWFP E	91	None	Interim riparian reserve; occasional density management, salvage	USDA and USDI (1993, 1994)
Federal Northwest Forest Plan, West of Cascades NWFP W	91–152	None	"Interim" but prevailing riparian reserve; occasional density management, salvage exceptions	USDA and USDI (1993, 1994)
Perennial non-fish-bearing streams				
WA FFR W	0–15	0–15		Washington Administrative Code (2006) [WAC 222-30-021(2)]
WA FFR E	0–15	0–15		Washington Administrative Code (2006) [WAC 222-30-022(2)]
WA DNR W	8	23		Washington State Department of Natural Resources (1997)
WA DNR E	0–15	0–15		Washington Administrative Code (2006) [WAC 222-30-022(2)]
OR Private small streams	0	0–3	Small streams have average annual water flow < 2 ft^3/s (cfs, ~57 l/s) or have drainage area < 81 ha	Oregon Administrative Rules (2006) [OAR 629-640-200 (6); OAR 629-635-0200]
OR Private medium to large streams	6	9–15	Streams with average annual water flow > 2 cfs (~57 l/s)	Oregon Administrative Rules (2006) [OAR 629-640-200 (2)(b); OAR 629-635-0310(1)(a); OAR 629-635-0200]
OR NW/SW State lands	8	23: inner zone; 21: outer zone	Applied to at least 75% of reach on small streams	Oregon Department of Forestry (2001)
CA FPR	None	15–30	Class II streams. Side slope dependent; Minimum 50% total canopy retention (overstory and understory combined)	Young (2000)
NWFP E	46	None	Interim riparian reserve; occasional density management, salvage	USDA and USDI (1993, 1994)
NWFP W	46–76	None	Interim riparian reserve; occasional density management, salvage	USDA and USDI (1993, 1994)
Seasonal non-fish-bearing streams				
WA FFR W	0	9	Equipment limitation zone only	Washington Administrative Code (2006) [WAC 222-30-021(2)(a)]
WA FFR E	0	9	Equipment limitation zone only	Washington Administrative Code (2006) [WAC 222-30-022(2)(a)]
WA DNR W	0	None	RMZ protection provided where necessary for aquatic system and in unstable areas (interim strategy)	Washington State Department of Natural Resources (1997)
WA DNR E	0	9	Equipment limitation zone only	Washington Administrative Code (2006) [WAC 222-30-022(2)(a)]

Table 4 (*Continued*)

Management system	No cut buffer (m)	Management zone (m) with timber harvest allowed	Comments	References
OR Private medium to large streams	6	9–15	Streams with average annual water flow > 2 cfs (∼57 l/s)	Oregon Administrative Rules (2006) [OAR 629-640-200 (2)(b); OAR 629-635-0310(1)(a); OAR 629-635-0200]
OR NW/SW State lands Small streams	0–8	23–30: inner zone; 21: outer zone	Applied to at least 75% of reach; small streams have average annual water flow ≤ 2 ft^3/s (cfs, ∼57 l/s) or have drainage area < 81 ha	Oregon Department of Forestry (2001), Oregon Administrative Rules (2006) [OAR 629-635-0200]
OR NW/SW State lands medium to large streams	8	23: inner zone; 21: outer zone	Applied to at least 75% of reach; Streams with average annual water flow > 2 cfs (∼57 l/s)	Oregon Department of Forestry (2001)
CA FPR	None	None	Class III streams. Side-slope dependent; Minimum 50% understory cover retention	Young (2000)
NWFP E	30–34	None	Interim riparian reserve; occasional density management, salvage	USDA and USDI (1993, 1994)
NWFP W	30–76	None	Interim riparian reserve; occasional density management, salvage	USDA and USDI (1993, 1994)

NA = not applicable. Conversion from English units in original literature to metric units are shown to nearest m. Management zones indicate widths of managed areas, not distances from stream (as shown in Fig. 2).

Other federal land management agencies, in particular the U.S. Bureau of Land Management (BLM), have operated under less explicit mandates for biological conservation than the U.S. Forest Service. In recent decades, however, the potential consequences of endangered species listing for wide-ranging terrestrial and freshwater taxa (some amphibians are candidate species for federal protection) prompted the unification of once-varied standards and practices under the umbrella of regional management frameworks. The Northwest Forest Plan, covering federal forest lands and waters within the range of the northern spotted owl (*Strix occidentalis*), is the most important of these (USDA and USDI, 1994). Its importance stems from the unprecedented convening of a multidisciplinary scientific team, known as the ''Forest Ecosystem Management Science Assessment Team'' (FEMAT) that developed science-based recommendations for uniform conservation measures for national forests and BLM lands within the range of the northern spotted owl (USDA and USDI, 1993). The FEMAT process resulted in the consideration and integration of a far broader scope of values, processes, and mechanisms of impact when riparian management rules were adopted in the Northwest Forest Plan than had previously been considered in federal management plans and project assessments. FEMAT stream protection guidelines extend beyond shade retention and filtration of sediment and nutrients, to the explicit consideration of long-term recruitment of coarse down wood to channels and soil surfaces, downstream transport of both wood and sediment to off-site areas, trophic sources from riparian habitats to aquatic food webs, and the effects of vegetation and vegetation management on riparian microclimate. Spence et al. (1996) lent further scientific support for the FEMAT approach and recommended the National Marine Fisheries Service adopt it in the development of Habitat Conservation Plans and other biological restoration and recovery measures.

The ecosystem approach embodied in the Northwest Forest Plan Standards and Guidelines established a benchmark for riparian conservation rules that no other agency or industrial landowner has yet approached (Fig. 2). Riparian reserves, the areas of restricted harvest adjacent to waterways and stream channels, extend from ∼30 m (100 ft) to more than 90 m (>300 ft) lateral to the stream channel on both sides of the stream. Reserve widths are framed in terms of site-potential tree height, the height a dominant mature tree would attain on a given site. Fish-bearing streams are given the widest reserves – the greater of 2 site-potential-tree heights or ∼90 m (Fig. 2a), while seasonally flowing non-fish-bearing streams have the narrowest widths – the greater of 1 site-potential-tree height or ∼30 m (Fig. 2c). Accounting for inherent differences in tree growth potential, prescribed riparian reserve widths are wider for more mesic forests west of the Cascade crest than for eastside, more xeric forests. While some forest management activities including tree harvest for density control or salvage are not absolutely prohibited in riparian reserves, they can only occur following an extensive assessment of their potential impacts with respect to a list of Aquatic Conservation Strategy objectives (USDA and USDI, 1994). Objective nine states: ''Maintain and restore habitat to support well-distributed populations of native plant, invertebrate and vertebrate riparian-dependent species''.

State and private forest managers are less clearly mandated than federal agencies to observe biological conservation objectives, but they are obligated to ensure that permitted or recommended practices meet the intent of the Clean Water Act, and under various treaties with Native American tribes, ensure the natural resource conditions necessary to sustain fishing, and hunting, and other uses or values practiced by indigenous peoples. Federal listings of salmon, bull trout (*Salvelinus confluentus*), and other fishes under the Endangered Species Act have in the past decade increased scrutiny of state-enforced forest practices laws, the authority under which most private forest management is conducted. Hence, more recent attempts to reform state and private forest practices rules, such as the Washington Department of Natural Resources' Forest Practices Habitat Conservation Plan (Washington State Department of Natural Resources, 1997; Bigley and Deisenhofer, 2006), have moved closer to the biologically based standards set by FEMAT.

Under this Habitat Conservation Plan in Washington, riparian management zones associated with fish-bearing streams are delineated into three sub-zones aligned as adjacent bands along streams: the uncut near-stream core, and the managed transitional inner and upslope outer zones (Fig. 2; Table 4). Conceptually, each zone provides different levels of riparian resources and functionality. The core zone is of fixed width and management activity with tree harvest limited to road construction for stream crossings and the creation and use of yarding corridors. Cut trees can only be removed from the core zone if coarse down wood targets are already met (a rare condition because of past management practices). The widths of inner and outer zones vary by stream width, site productivity class, and the type of management selected by the landowner. Harvest activities in the inner zone are limited to a set of specified silvicultural options and can be undertaken only if projected stand development meets threshold desired future conditions for tree density, basal area per acre and proportion of conifer species. Timber harvest in the outer zone is generally allowed subject to the retention of a specified minimum density of riparian trees.

In contrast, California Forest Practice Rules do not stipulate a mandatory no harvest zone adjacent to fish-bearing or non-fish-bearing streams typical of headwater forests. Instead, Watercourse and Lake Protection Zones are defined based on stream width and near-stream topography, with steeper slopes requiring wider protection zones (Young, 2000). Harvesting within these zones of streams supporting fish or providing habitat for non-fish aquatic species (Class I and II streams) is restricted to the retention of a specified percentage of overstory and understory canopy cover rather than a minimum residual tree density or basal area, with the additional requirement of retention of a minimum density of large trees within 15 m of the channel. In small streams lacking evidence of aquatic life (Class III), the minimum canopy cover restriction can be met by understory vegetation alone. Furthermore, debate exists over what constitutes aquatic life (currently aquatic invertebrates are not recognized as such in California) which results in many likely Class II channels being mis-classified and receiving only Class III protections (H. Welsh, personal observation). A second problem involves the timing of efforts to establish the presence of vertebrate life in headwater channels. This often is done in the dry season when tributary flows go subsurface, and aquatic amphibians disappear into the substrates of the hyporheic zone (Feral et al., 2005) to await fall rains. This unfortunate timing results in the misclassification of streams which erroneously puts their fauna at risk.

Misclassification of stream types also was documented in Washington due to an over-reliance on maps derived from Geographic Information Systems and remote sensing: 23% of fish-bearing streams were misclassified as having no fish; 39% of non-fish-bearing streams were not identified on maps (Rashin et al., 2006). While this has led to adaptive management of the stream identification process, Rashin et al. (2006) suggested ground truthing would be needed to ensure accuracy.

Consistent across the riparian protection schemes outlined are (1) a greater width of protection zone for larger streams and fish-bearing streams; (2) decreasing intensity of management activity allowed with increased proximity to the stream; and (3) vegetation retention designed to provide near-stream shade, sediment filtration, and bank stability. However, examination of Fig. 2 and Table 4 clearly reveals that differences among management jurisdictions are largest when it comes to headwater streams that are not fish-occupied. These streams are afforded narrower protective buffers than are large, fish-bearing streams regardless of whether their dry-season flows are permanent or intermittent. Kondolf et al. (1996) and Welsh et al. (2000) have assailed the logic of narrower buffers given that steeper, headwater streams occupy the position in the stream network where the majority of sediment and nutrient transfer from land to water occurs. Forman (1995) also considered wider buffers in headwater streams a more prudent approach due to the significant downstream benefits they contributed. Nevertheless, narrow "buffers" within which extensive logging can occur remains the current standard on private forest lands throughout the region (see Fig. 2, Table 4). Questions persist about whether narrow buffers provide sufficient moderation of microclimate, habitat diversity, and transfers of energy and matter to support non-fish aquatic and riparian biota, particularly sensitive frogs and salamanders, whose abundance is often greatest upstream of fish-bearing waters and whose adult stages sometimes forage hundreds of meters upland from the immediate stream margin.

2.2. Riparian management, stream temperature, and microclimate

Harvesting of riparian vegetation has been repeatedly shown to result in alterations of stream temperature regime including increased average and maximum temperatures and increased diurnal variation (Johnson and Jones, 2000; Herunter et al., 2004; Wilkerson et al., 2006). Furthermore, removal of stream shade can lead to an earlier seasonal occurrence of stream temperature extremes, possibly as a result of changes in the relative influences of incident solar radiation and seasonal low flow in determining maximum stream temperature (Johnson and Jones, 2000; Wilkerson et al., 2006).

The magnitude of stream temperature response to harvest will vary with the amount of stream shade retained, the intensity of upslope harvest, and time since harvest. Complete removal of stream shade from headwater streams may result in temperature increases of as much as 5–13 °C (Johnson and Jones, 2000; Macdonald et al., 2003; Moore et al., 2005). Retention of buffers along headwater streams may result in negligible increase or as much as 5 °C increase in maximum stream temperature, depending on buffer widths and buffer density (Wilkerson et al., 2006; Jackson et al., 2001; Moore et al., 2005). Evaluating the effects of clearcutting adjacent to intact buffers of 10 or 30 m width in British Columbia, Kiffney et al. (2003) observed increased stream temperatures of nearly 5 °C for the narrow buffers and increases less than 1.6 °C for 30 m buffers. In contrast, for headwater streams in Maine, Wilkerson et al. (2006) observed negligible increases in headwater stream temperature when buffers of 11 m width and >60% canopy cover were retained adjacent to clearcuts. In central British Columbia, partially harvested buffers (20–30 m wide) were less effective in stream temperature mitigation, with high and low retention buffers associated with 1–3 and 2–4 °C increases, respectively (Herunter et al., 2004; Macdonald et al., 2003). Dense deposits of logging slash over the steam channel has been observed to prevent a stream temperature increase following clearcutting (Jackson et al., 2001) further emphasizing the importance of shade in regulating stream temperature response. Increased stream temperatures following harvest have been observed to persist for 5 years (Macdonald et al., 2003) and in excess of 15 years (Johnson and Jones, 2000). Stream temperature recovery following harvest is likely driven by development of riparian vegetation with rates of recovery being potentially greater in mesic forests such as the Coast Range of Oregon and Washington than more xeric forests such as those of the east-side Cascade Range or the Siskiyou Mountains of southern Oregon. Even with buffering, stream temperature recovery may be delayed if the buffers undergo post-harvest density reductions due to windthrow (Macdonald et al., 2003) or other disturbances.

While incident solar radiation may be the primary driver of stream temperature response to harvest of riparian vegetation (Brown and Krygier, 1970), hydrological influences can be strong, particularly in headwater streams having seasonal low flows and low depth to surface-area ratios. At the catchment scale, harvest may decrease transpiration and result in a transitory period of higher summer minimum flows lasting a few years to more than a decade (Moore and Wondzell, 2005). While increased minimum flows may tend to mitigate stream heat loading, there has been some suggestion that upslope harvests, particularly clearcutting, may increase the temperature of sub-surface flows entering headwater stream channels (Brosofske et al., 1997) leading to increased stream temperature. Furthermore, changes in vegetation composition following harvest may alter flow patterns relative to pre-harvest; conversion from conifer to hardwood riparian vegetation may result in lower summer minimum flows (Moore and Wondzell, 2005).

There is substantial interest in the potential for downstream ‘‘relaxation’’ (Ice, 2001) of stream temperature responses to harvest. While increases in stream temperature can be cumulative, thermal pollution is not conserved and stream temperature is constantly moving toward equilibrium with the surrounding environment (Ice, 2001). Thus, streams passing through harvest units may cool if they subsequently pass through shaded, cooler conditions. Zwieniecki and Newton (1999) reported a 2 °C stream temperature decline within 300 m downstream of harvest units in western Oregon. Wilkerson et al. (2006) observed complete temperature recovery within 100 m of the harvest unit boundary on headwater streams in Maine. Story et al. (2003) concluded that thermal recovery of headwater streams in central British Columbia was due predominantly to ground water inflow. While stream temperature relaxation may be a common phenomenon, Ice (2001) concluded that it is more ecologically efficient to use shade to protect stream water from temperature increases than it is to cool water that has been warmed.

While buffers have been demonstrated as having the potential to mitigate harvest effects on headwater stream temperatures, it is important to consider the potential biological consequences of even small changes in thermal regime. Thermal tolerances of some fish and amphibian species have been determined. For example, Huff et al. (2005) determined the thermal tolerances for 16 aquatic vertebrates in four ecoregions of Oregon. Stream temperature minima and maxima differed not only among species, but also varied among subpopulations of a species occurring in different ecoregions (Huff et al., 2005). Beyond the general impacts of increased stream temperature, seasonal shifts in stream temperature extremes may negatively impact organisms at sensitive stages of development (e.g., larval development), or may negatively alter behaviors, such as triggering an early migration of anadromous fishes (Macdonald et al., 2003). While Kiffney et al. (2003) found 30-m buffers adequate to mitigate stream temperature effects, they observed significant changes in periphyton communities, and thus primary productivity, in buffered streams. Breakdown of leaf litter by microbes and invertebrates in headwater streams was little influenced by temperature variation, but rather was more strongly influenced by water chemistry (Rowe et al., 1996). Welsh et al. (2005b) demonstrated that changes in water temperatures, sufficient to change the composition of the in-stream vertebrate assemblage (including the loss of coho salmon [*Oncorhynchus kisutch*]), resulted from the removal of more than 15% of the forest cover in small tributary basins in northern California.

Recent riparian microclimate research has addressed the question of how forest management practices influence riparian microclimate. Alteration of stream temperature has been a primary concern, however retention of interior forest conditions along streams, including streamside and riparian zone microclimates, is gaining recognition as a riparian management objective. The question is often framed as ‘‘what measures must be taken to prevent disturbance of stream and riparian microclimates considered critical to ecosystem function?’’ With our new knowledge of trans-riparian stream effects, this could be rephrased to ask how edge effects from forest management interact with stream effects? The studies in Table 1, with the exception of Welsh et al. (2005b), examined combinations of forest harvest and riparian buffering on microclimate. In all cases, some degree of microclimate moderation was derived from streamside vegetation.

The degree of microclimate moderation afforded by buffers can depend on the reference condition against which impacts are assessed. Meleason and Quinn (2004) assessed buffer width effectiveness as the extent to which air temperature in the buffer was decreased relative to the upslope clearcut forest. Hence, this does not address change from a pre-harvest condition, potentially highly relevant to forest biota. Anderson et al. (2007), S. Chan et al. (unpublished data [OSU]), and Hagan and Whitman (2000) evaluated stream-center and riparian-buffer microclimates in harvested stands (clearcut or thinned) relative to similar locations in unharvested stands. Brosofske et al. (1997) and Dong et al. (1998) assessed alteration of stream center microclimate relative to pre-harvest stream-center conditions and to upslope microclimates of nearby intact late-seral forests and clearcut openings.

Clearcut harvesting clearly alters microclimates in the harvested area, but its influence on buffered riparian areas is not straightforward. Meleason and Quinn (2004) found that while air temperatures at the center of buffers were clearly lower (∼3 °C) than those of adjacent forest openings, buffer maximum daily temperatures differed only marginally (∼0.3 °C) for buffers of 5- and 30-m widths. Rykken et al. (2007a) examined buffers of approximately 30-m width and found that summer maximum daily temperatures within the buffers averaged about 35% lower than in adjacent clearcuts and about 10% higher than those in intact late-seral forests. Brosofske et al. (1997) and Dong et al. (1998) observed up to a 4 °C increase in stream-center air temperature, with clearcutting adjacent to buffers ranging from 0 to 72 m wide; they did not collect microclimate data within the buffers.

Increasingly, thinning of young, second-growth stands is being prescribed on federal lands in the Pacific Northwest to meet compatible objectives for watershed restoration and economic revenue. Evaluating a range of buffer widths from approximately 6–70 m, Anderson et al. (2007) found that summer mean daily maximum air temperature at the stream center was minimally affected by upslope thinning (∼1 °C increase relative to unthinned reaches) when unthinned buffers of at least 15 m width were retained. For narrower, streamside retention buffers (∼6 m average width) prescribed primarily for streambank stabilization, stream-center air temperatures increased ∼4 °C, reflecting a breakdown of the near-stream air temperature gradient. In thinned stands, stream-center relative humidity decreased an average of 18% with narrow, streamside retention buffers and only about 5% when buffers 15 m or wider were retained. No discernable differences existed for either air temperature or relative humidity measured at stream center among buffers 15 m and greater in width. Within buffers, maximum air temperatures averaged 0–2.5 °C higher and relative humidity averaged from 1% higher to 27% lower than at stream center, depending on buffer width and thinning intensity. These findings illustrate that less intensive thinning harvests that retain a substantial proportion of the pre-harvest stand density and canopy cover have less impact on stream and riparian microclimates than do more intensive regeneration harvests.

The predominant mechanism by which buffers influence stream and riparian microclimate is presumed to be shading—the blocking of solar radiation from reaching the stream channel or forest understory. However, other factors also influence riparian microclimates, including local topography, hydrology and macroclimate. Many of the studies summarized here were conducted at multiple sites having different local site characteristics. Interestingly, in general these studies have not explicitly addressed the influence of site variation in their analyses (but see Welsh et al., 2005b). Danehy and Kirpes (2000), observing wide variation among sites in the strength of correlation between percent canopy cover and relative humidity gradient, concluded that local topography could override shade effects on relative humidity. They attributed some of this among-site variation to the steepness of near-stream topography—stronger microclimate gradients were associated with streams having steep side slopes and weak gradients were associated with streams having shallow valley cross sections. Furthermore, Danehy and Kirpes (2000) suggested that in the drier areas east-side of the Cascade Range crest, seasonal limitations in moisture may result in narrower zones of stream-influenced microclimate, although the zone of stream influence as represented by relative humidity seemed to be very similar to that reported by Anderson et al. (2007), S. Chan et al. (unpublished data [OSU]) and Rykken et al. (2007a) for western Oregon.

At this time, given the studies that have been conducted and site-specific contexts of the results, it is difficult to make prescriptive buffer width recommendations to retain riparian microclimates. A 300-m buffer was considered by Brosofske et al. (1997) as sufficient to maintain relative humidity gradients typical of uncut forests. However, following harvest, they did not sample microclimate within the buffer and therefore could not detect either the extent of a near-stream steep gradient (stream effect) or the distance from stream at which gradients depart from those of the typical uncut condition. In contrast, for retention of streamside riparian microclimate and invertebrates, Rykken et al. (2007a,b) proposed a narrower 30 m buffer when clearcut harvest occurs upslope because they found a microclimate "stream effect" counterbalanced the edge effect from the cut boundary at this closer distance to the stream (Rykken, 2004). Due to the stream effect, again evident by the steep near-stream temperature and relative humidity gradients, this narrower buffer zone was concluded to be sufficient to preserve microclimate conditions within 5 m of streams. However they did not address alternative buffer widths nor sample intensively across the trans-riparian gradient to show how stream and edge effects interacted. With upland forest thinning rather than clearcutting, it is possible that a narrow buffer width might preserve microclimates at streams, but this is uncertain and may be variable. Anderson et al. (2007) suggested that a 15 m-wide buffer may retain stream center conditions (see above), but the distribution and replication of microclimate data-loggers used in the study was not designed to characterize the spatial patterns of microclimate about the buffer-upslope edge, and therefore the study does not fully address the spatial extent to which stream effects counterbalance edge effects. Given limitations of the sampling designs in the studies reviewed, we can conclude that relatively narrow buffers (relative to those of the Northwest Forest Plan, in particular) can be effective in maintaining stream center microclimate conditions and therefore the steep near-stream microclimate gradients that may extend 10–20 m from streams in intact stands (Anderson et al., 2007; see above microclimate discussion). However, we have few data for predicting the countervailing spatial extent of upslope harvest influences on microclimates within buffers. In general, our understanding of trans-buffer microclimate gradients must be improved by sampling at a higher spatial resolution with spacing among sensors sufficient to quantify non-linear trends across ecotones associated with both stream-buffer and buffer-upslope edges. Such potential research is further complicated by the likelihood of context-specific results on riparian microclimates from both site conditions and silvicultural practices. Additionally, to address our uncertainty of species-specific effects across trans-riparian zones, it would be valuable to track species responses simultaneously.

2.3. Riparian management and biodiversity

Management to consider the ecological needs of stream–riparian dependent biota is not a new concept. However, for flora, impacts of forest management on riparian vegetation community structure are not well documented (Pabst and Spies, 1998). Hibbs and Bower (2001) found little evidence of vegetation community differences between buffer strips adjacent to clearcuts and similar near-stream zones in unharvested reaches. The buffer configuration evaluated was limited to a single strip of trees adjacent to the stream. Additional efforts will be needed to evaluate the effectiveness of buffer areas as riparian habitat (i.e., considering within-buffer vegetation responses) to varying intensities of harvest and for various widths of buffer, especially considering the diversity of assemblages identified by Pabst and Spies (1998) and Sheridan and Spies (2005). For animals and the physical habitat attributes on which they depend, numerous studies have investigated riparian associations and called for protection of riparian habitat from logging and other management disturbances (e.g., Table 5). Headwater streams that constitute habitat for endemic amphibians such as tailed frogs and torrent salamanders have been recognized as receiving insufficient protection for almost two decades (e.g., Bury, 1988; Bury and Corn, 1988; Corn and Bury, 1989; Dupuis and Steventon, 1999; Welsh et al., 2000; Welsh and Lind, 2002; Ashton et al., 2006).

To retain riparian fauna, recommendations for buffer widths have spanned 6 to >90 m, depending upon the upslope management scenario and site-specific conditions (Table 5). The narrower 6-m estimate stems from moderate thinning of young (~40–50 year) managed stands conducted outside of small headwater streams, where instream fauna were not negatively affected in years 1 and 2 post-harvest (Olson and Rugger, 2007). The researchers cautioned that lag effects may occur in subsequent years, and to hedge uncertainties, suggested a mix of buffer widths be used within a planning area. Additionally, this 6-m buffer resulted in a 4 °C increase in summer daily maximum air temperature at streams, hence microclimate was not retained by this small no-cut buffer with thinning upslope (Anderson et al., 2007), although during this timeframe (summer, ~4 pm) amphibians are not surface active. The larger estimate, >90 m, was derived by examining distances from streams used by aquatic salamanders and adding an additional 50 m to buffer edge effects to the complex of aquatic and terrestrial habitats they require (Crawford and Semlitsch, 2007). Wider buffers likely would be needed for retention of trans-riparian conditions relative to those occurring in unmanaged stands, and for riparian biota that venture farther upslope (Table 3). Several studies recommend buffers of 30 to >45 m for riparian obligate birds and amphibians (Table 5). Movements of stream-breeding amphibians to 100, 200, and 400 m upslope suggest increased distances and inter-stream connectivity need consideration. Hence, both a

Table 5
Stream riparian buffer width recommendations to retain various species or conditions (primary focus on PNW forests)

Width (m)	Rationale	Reference
6–76	Retains stream and bank amphibians (small streams with thinning)	Olson and Rugger (2007)
20	Contained 80% of *Rhyacotriton*, *Plethodon dunni* and *Dicamptodon tenebrosus*	Vesely and McComb (2002)
27	Retains 95% amphibian assemblage in southern Appalachian streams, North Carolina	Crawford and Semlitsch (2007)
42.5	Needed for 1 species	
92.6	Additional 50 m to buffer edge effects.	
30	Retains frogs—Australia	Lemckert and Brassil (2000)
30	Retains riparian invertebrates; retains riparian microclimates (small streams with clearcuts)	Rykken et al. (2007b)
30	Primary activity of aquatic trichoptera (may go as far as 200 m upslope)	Collier and Smith (1998)
30–40	Retains most riparian-associated amphibians, unlikely to retain upland species or vagile pond breeders such as northern red-legged frogs and roughskin newts	Vesely (1996)
30	Most riparian birds	Pearson and Manuwal (2001)
>45	Black-throated gray warbler	
>40	Forest-associated bird species in Oregon Coast Range headwaters	Hagar (1999)
45–300	Retains riparian microclimates (with clearcuts)	Brosofske et al. (1997)
≥45	Retains birds with low species turnover	Pearson and Manuwal (2001)
>46	Retains stream amphibians (small streams with clearcuts)	Stoddard and Hayes (2005)
70–90	Full complement of riparian-stream linkages, such as down wood, litter, bank stability, microclimate (one site-potential tree height)	Young (2000)

stream buffer approach to retain aquatic connectivity and an upland inter-stream approach for terrestrial connectivity could be considered simultaneously. It is largely unknown how these animals might respond to different zones of riparian buffer management; alternative buffer management designs require further investigation relative to the different species.

The extent, pattern, and duration of connectivity between stream–riparian patches needed to minimize fragmentation effects in Pacific Northwest forests are unknown, but these features undoubtedly vary among species. Given the affinity of stream–riparian amphibians for habitat above the fish-inhabited reaches of the stream network, it would seem likely that connectivity between stream–riparian habitat fragments would be more ecologically effective for amphibians above the fish-bearing zone (i.e., across headwaters) than connectivity along interconnecting higher-order stream reaches that are inhabited by fish. Hence, with the sum of what we know about the biology of headwater stream and riparian species, a precautionary approach might be warranted to maintaining connectivity at multiple spatial and temporal scales, such as within and between subdrainages, and between larger basins (e.g., Richards et al., 2002), and over short and long timeframes across juxtaposed lands.

2.4. Spatial pattern of headwater reserves

Key management actions or design elements to minimize fragmentation effects on Pacific Northwest stream–riparian amphibians are to keep fragments as large as possible, and to maintain connectivity of habitat fragments by retaining some contiguous unlogged habitat between them (e.g., Richards et al., 2002). These two actions correspond to two of the ‘‘critical elements of effective and biologically based management plans for amphibians that consider both population and landscape processes,’’ namely protection of terrestrial buffers around wetlands and protection of the integrity of ecological connectivity (Semlitsch, 2000). Specific buffer distances have been proposed for wetland-breeding amphibians (Semlitsch and Bodie, 2003) and stream–riparian associates (Table 5), whereas specific guidelines for maintaining connectivity have not. Given the penchant for some stream-dwelling amphibians to move longitudinally within a stream network (e.g., Hayes et al., 2006), one design for maintaining connectivity would be to extend some of the buffers protecting adjacent headwater streams up to the ridgeline where they would join (Sheridan and Olson, 2003) (Fig. 3c and d). Bury (1988) similarly proposed such a patch reserve at headwaters for amphibians,

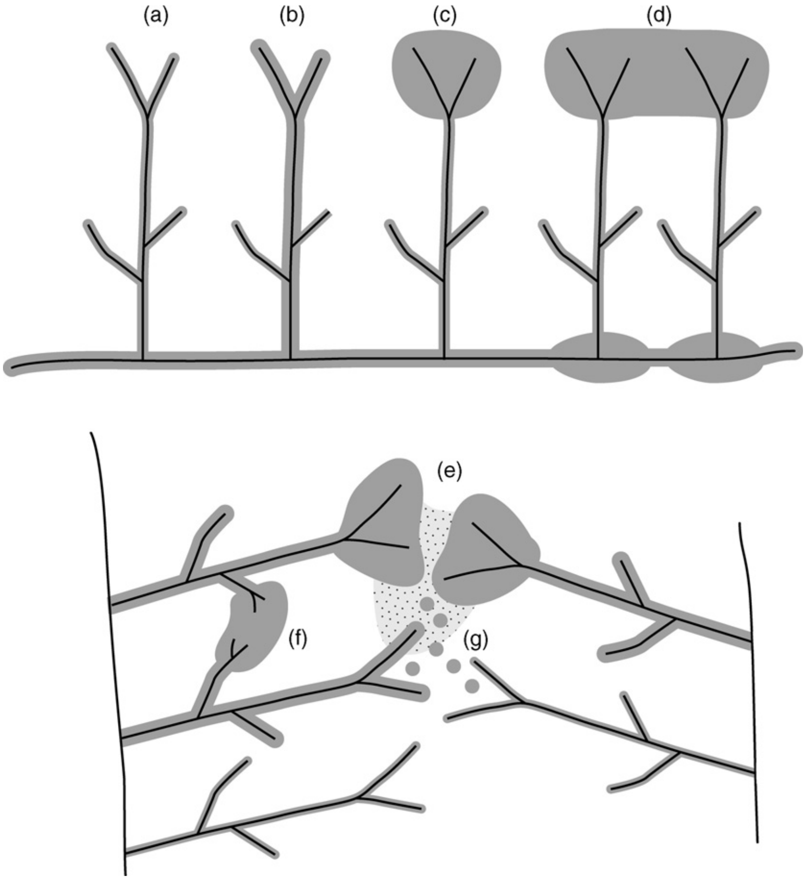

Fig. 3. Riparian management considerations for retention of headwater amphibians (‘‘spaghetti and meatballs’’ approach): (a) narrow buffer zone might be used to protect water quality and some instream habitat components where headwater amphibian occurrences or habitat quality are low; (b) wider buffer 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 watersheds.

and also considered patches along streams to be a valuable approach. Patches or discontinuous patches along streams also were proposed by Bury and Corn (1988). Cissel et al. (1998) designed headwater reserves for this purpose, and included patch reserves downstream at tributary junctions for combined biodiversity and habitat functions. Another approach would be to protect the entire upper portion of adjacent stream watersheds on both sides of a ridgeline (Fig. 3f). The dimensions to make such connections effective are not known; however, in principle, a corridor between patches is anticipated to reduce fragmentation, facilitate dispersal and increase the effective size of populations. While no-cut connections could provide optimum habitat, partial harvest (dispersed or aggregated trees) across ridgelines might provide habitat conditions for dispersal of many species (Fig. 3e and g).

Carey (2006) provided a list of tools to manage upland forests for biocomplexity objectives, retaining structural forest features and softening the impact of harvest activities. Thinning may not reduce occupancy or abundance of numerous taxa (e.g., plants, arthropods, mollusks, amphibians: Wessell, 2005), but Karraker and Welsh (2006) found body condition of terrestrial salamanders was significantly reduced on thinned stands. Knapp et al. (2003) cautioned against implementation of low-intensity harvest everywhere, and to minimize effects on salamanders they proposed concentrating high intensity harvest in small areas, and leaving uncut areas as likely optimum habitat (the considerable impact of an extensive road network necessary to implement landscape-extensive thinning regimes must also be considered). Patch reserves or leave islands in such a logged matrix could serve as habitat refugia or ‘‘stepping stones’’ between drainages (Fig. 3g). Leave islands may retain some species, provide wood and structure (Matveinen-Huju et al., 2006) including species associated with late-successional forest conditions (Wessell, 2005). Furthermore, interior forest microclimates were preserved in 0.4-ha circular leave islands within young managed stands in western Oregon (Wessell, 2005). Similar to Olson and Rugger (2007) suggesting a mix of buffer widths to be applied across drainages, Lowe and Bolger (2002) proposed a mosaic of conditions be implemented upslope. These are conceptual designs at present, and research is needed to understand the population-level implications of connectivity designs in addition to combined buffer-and-connectivity designs at subdrainage and basin scales. This ‘‘spaghetti and meatball’’ approach of linear stream buffers (spaghetti reserves) and patch reserves (meatballs), either along streams or upslope, warrants testing.

2.5. Target species management

While maintaining riparian forest biodiversity is of paramount concern, it is well recognized that forest resources such as timber result in conflicting land management priorities for managers. How can we sustain biota and commodities? We suggest an approach including target-species management in headwaters, where key taxa and their priority habitats are identified and managed for species persistence. This type of prioritization of areas for management focused on the needs of a set of target species is gaining broader use in other systems. The concept acknowledges that to maintain a sensitive species well-distributed across its range, all areas may not need a conservative approach ensuring a low risk to species extirpation. For example, under the federal Survey and Manage program (USDA and USDI, 2001), all known sites may not warrant species-focused management for those with sufficient distribution to be considered ‘‘uncommon’’ rather than ‘‘rare.’’ For uncommon species, sites could be prioritized for management. For headwater amphibians, this notion has been expressed by several researchers. In British Columbia, Dupuis and Steventon (1999) suggested priority management areas be identified for tailed frogs along creeks with higher frog abundances and coarse, stable substrates, a habitat attribute associated with their occurrences. Similarly, larger-order non-fish-bearing basins could be regarded as higher priority watersheds for coastal tailed frogs (Hayes et al., 2006). For many stream amphibians in the Pacific Northwest, physical habitat features of streams such as substrate composition are strong predictors of occurrence or abundance (e.g., torrent salamanders [*Rhyacotriton*] and tailed frogs [*Ascaphus*]: Wilkins and Peterson, 2000; Adams and Bury, 2002; Russell et al., 2004), and should be considered if management areas are prioritized. Coastal tailed frogs have patchy distributions with optimal habitat conditions in late-seral forests (e.g., Welsh and Lind, 2002), and may need species-specific consideration for management across their range, which is broadly subject to timber harvest activities. Also, southern torrent salamanders, *R. variegatus*, have been found more frequently in older forest sites and may rely on microhabitat and microclimate protection (Welsh, 1990; Welsh and Lind, 1996). Bury and Corn (1988) and Bury et al. (1991) suggested that if tailed frogs or torrent salamanders were present, then protection measures should be implemented, although Bury and Corn (1988) felt the entire stream length might not need protection for the purpose of maintaining these species. Across the Pacific Northwest, these two taxa, tailed frogs and torrent salamanders, have gained concern among researchers, and are prime candidates to be considered in our target-species headwater protection scheme. At local scales, other species may be identified as targets for management, such as those with restricted distributions like the Van Dyke’s or Coeur d’Alene salamanders (*P. vandykei* and *P. idahoensis*, respectively).

While our focus in this analysis has been on headwater amphibians, this approach may be suitable for other species or habitat areas. For example, the stream–riparian dependent western pond turtle, *Clemmys marmorata*, nests in uplands, sometimes hundreds of meters away from water (Reese and Welsh, 1997; Blaustein et al., 1995). Nesting sites may be aggregated and the same area used every year, so a priority management area could be designated for this species’ habitat.

If not every stream reach or stream is managed for these endemic headwater species, how many priority management areas are needed? A larger scale than the stream reach or stream needs consideration. We suggest that it might be practical and effective for species distributions to be managed for persistence at the scale of 6th-field watersheds (i.e., 6th code hydrologic units; catchment areas defined by an integrated stream network and typically ranging from ~4000 to 16,000 ha in the Oregon Coast Range; K. Burnett, personal communinication [U.S. Forest Service]), as management designs at this scale can incorporate approaches along and between streams. Ideally, multiple areas within a 6th-field watershed could be identified to anchor headwater species and processes. As 6th-field watersheds are aggregated, larger basin connectivity can be addressed. Moreover, as genetic information becomes available for headwater species, the effectiveness of this approach can be tested.

This approach requires knowledge of species' distributions, to focus management to areas of high species abundances and/or high habitat quality, or both. This requires inventories for targeted species preceding development of priority species management areas. The costs of screening an area for a target species would be offset by the benefit to the land manager of the likely lifting of timber harvest restrictions over a portion of the watershed, while ensuring provision of within- and among-site protections to result in well-distributed populations of target species across the landscape. Across watersheds, the ''spaghetti and meatball'' toolbox can be implemented, potentially including narrow stream buffers to retain bank stability and stream habitat attributes (Fig. 3a), wider buffers for additional stream habitat and slope stability attributes and some riparian refugia (Fig. 3b), patch reserves at headwaters or along streams (Fig. 3c and d), and connections by full or partial forest retention across ridgelines of neighboring basins (Fig. 3e–g).

2.6. Balancing conflicting management priorities

In conjunction with a target-species approach, riparian and landscape design tools can be ordered by those that provide more conservative management approaches for aquatic–riparian dependent species, and those that are more geared to balancing conflicting forest management priorities. This dichotomy of designs has been suggested by the U.S. national Partners for Amphibian and Reptile Conservation, in their Habitat Management Guides for the southeastern U.S. (Bailey et al., 2006). A 2-tiered management design reiterates the idea that not all species-sites have to be conservatively managed. However, if designs can be developed at larger spatial scales, well-distributed populations and connectivity of managed sites can be addressed, perhaps by a mix of approaches. These guides are developed to assist private, industrial, state, federal or other land owners in managing for herpetological species across diverse habitats nationwide. Similar guidelines are in development for the rest of the country, and the northwest Habitat Management Guides (D. Pilliod and E. Wind, unpublished data) are expected to be available in 2007.

Conservative and forestry-compatible considerations for stream–riparian dependent species in the Pacific Northwest forests are included in Table 6. Conservative approaches are geared to protect the stream and riparian habitats used by obligate species and incidentally used by facultative species. Hence a wider riparian buffer width is listed, 40–150 m, to retain riparian and upland habitats used by stream–riparian dependent fauna. These wider buffer areas might also take the form of ''bulges'' or patch reserves placed along streams, perhaps located at headwaters or tributary junctions, used in a forestry-compatible approach. These forested buffers might be management zones rather than no-entry reserves, if habitat conditions such as forest structural elements or stream hydrology can be shown to warrant some form of restoration management, or if there is thought to be limited risk to key taxa or conditions from the proposed management activities. Long-term large down wood recruitment is a particular concern for second-growth stands in these buffers. Included in Table 6 is a recommendation for avoidance of chemical applications and road building in riparian areas, due to their various adverse effects on amphibians, including direct mortality (e.g., deMaynadier and Hunter, 1995). Stream road crossings may disrupt movements of animals over the road and along the stream or riparian area; for example, Sagar et al. (2007) found larval coastal giant salamander movements instream were affected by culverts.

Approaches to balance conflicting species and timber production priorities may be a mix of buffers widths, patch reserves of different sizes, and dispersed tree retention (Table 6). However, as larger spatial scales are considered, retaining species distributions can be an overarching objective, so an integration of approaches is likely needed, with target-species approaches helping to focus protective efforts in key locations for species of interest or concern. The benefit of this 2-tiered management approach to forest landowners would be reduced harvest restrictions on some lands while other areas would function to retain headwater taxa and likely other ecological services; specifically, this approach would be geared towards precluding downward population trends towards species-listing under the Endangered Species Act. This approach requires testing for its benefits to headwater target species as well as for its efficacy in retaining other forest ecological services, including other headwater species and functions.

Headwaters are not the only area in a forested landscape warranting concern for forest-dependent species. Biodiversity management areas could be consolidated as multiple species' distributions overlap. For example, as targeted headwater species' distributions overlap owl nest sites, botanical set-asides or other upland rare-species protection areas, these areas could be considered to serve multiple biological functions. Again, we suggest consideration of 6th-field watershed designs for multiple species management areas to anchor populations across landscapes. Central to this notion is that many species could also occur in the intervening matrix at some spatial or temporal scales, but the anchor areas would provide more optimal habitat conditions and likely source populations. A 6th-field watershed scale of habitat anchors could address many species connectivity issues, except those with extremely limited dispersal abilities, and could hedge uncertainties of natural or anthropogenic disturbances affecting local anchor sites.

Table 6
Stream riparian management considerations for biodiversity

Approach	Consideration	Example
Conservative	Maintain and restore the aquatic, riparian and upland systems simultaneously	1. Retain connectivity longitudinally and laterally from stream channels, into headwaters and across ridgelines. 2. Consider species with life histories transecting perennial-to-ephemeral channels, stream-to-riparian and riparian-to-upland systems
	Identify high priority areas for protection or restoration	1. Unique habitats 2. Areas with high disturbance potential 3. Areas outside the range of natural variation 4. Areas with unique species or species of concern
	Provide aquatic–riparian protection via delineation of entire subdrainage reserves, patch reserves and/or riparian buffers	1. 40–150 m (∼130–500 ft) buffers for aquatic and riparian habitat and species 2. Species specific considerations, e.g., turtle nesting sites may be >150 m from water 3. A mix of subdrainage reserves, patch reserves, and buffer widths may integrate local knowledge of habitats, or to hedge uncertainties
	Maintain or restore microhabitats	1. Large down wood 2. Interstitial spaces in substrates 3. Vegetation 4. Microclimates
	Maintain or restore natural hydrological conditions	1. Peak flow timing and extent 2. Flow duration
	Avoid chemical applications	1. Fertilizers 2. Herbicides and pesticides 3. Fire retardants
	Maintain stream continuity at road crossings	1. Avoid new road construction in riparian areas 2. Avoid pipe culverts with perched outlets at stream crossings
Forestry-compatible	At the landscape scale, consider connectivity of aquatic and terrestrial habitats in management plans	1. Stream channels to uplands 2. Headwaters to ridgelines 3. Corridors linking areas
	Consider applying a mix of riparian protections, such as different buffer widths or combined linear buffers with patch reserves in polygon shapes	1. 10 m (∼30 ft) for bank stability 2. 15–30 m (∼50–100 ft) for some water quality and aquatic habitat attributes 3. 40–100 m (∼130–330 ft) for aquatic/riparian-dependent species 4. Tiered and/or interspersed larger and smaller zones.
	Consider seasonal restrictions on management activities and ground disturbances in or near riparian areas	1. Timing to reflect the annual life cycle of activities of the resident species of concern (e.g., spring and fall amphibian activities)

Conservative approaches may be used when benefiting biodiversity is a primary objective. Forestry-compatible considerations include approaches to balance conflicting resource objectives, and may be used when biodiversity retention is secondary to other land use objectives.

2.7. Monitoring

Refinement of these management designs requires experimental manipulation or monitoring. For endemic headwater species, such as tailed frogs and torrent salamanders, it may be argued that conservative management approaches should be used until designs have empirical support for success in sustaining populations. Unfortunately, a call for research on these species has been made for two decades (Bury, 1983; Corn and Bury, 1989), and while significant new information is available on habitat associations, and many ideas have been forwarded as to how management might proceed, we have relatively little experimental or empirical knowledge about the efficacies of alternative management scenarios. Deployment of an experimental approach in a landscape-level test of the effectiveness of alternative prescriptions in headwater streams is currently underway in Washington State, with monitoring planned over several years post-implementation, but the result of this work will not be available for several years (M. Hayes, unpublished data). Measures of success for headwater amphibian species include validation that species occupancy persists or is increased across stream reaches within basins, relative abundances similarly do not show decreasing trends, and habitat attributes associated with these species are not showing patterns of degradation (sedimentation, water temperature, water flow, down wood recruitment). Field studies and genetic tools could be used to assess connectivity and population metrics in basins subject to different management regimes. It would also be important to determine that this approach does not result in additional risk to other species of concern or to key ecological functions of headwaters.

3. Conclusions

A new understanding of forest management designs to retain stream–riparian habitats and biodiversity is developing. We synthesize emerging data on riparian microclimates, efficacy of buffer widths for microclimate retention and species retention, spatial distributions of riparian-dependent fauna, and the need for upland connections between streams. The amphibians of the forested landscape of the Pacific Northwest are a particular concern, with declining population issues becoming more apparent regionally and globally. All 47 species occur in riparian areas over at least a portion of their range, 90% occur in forested habitats, about a third are stream–riparian obligate species, and a quarter are tied to headwaters. A new conservation approach outlined here targets selected species-of-interest for management designs along and among headwater stream reaches. A conservation approach for species persistence incorporates wider riparian management zones (40–150 m) and patch reserves along headwater streams to accommodate terrestrial life history functions of stream–riparian associated fauna, and habitat management in upslope forests to promote connectivity among drainages. A mix of buffer widths, 6–100 m, is suggested when timber management in forestlands is the dominant priority. Developing headwater habitat anchors at the spatial scale of 6th-field watersheds offers a design for connectivity of populations across forest landscapes. Piggy-backing protections of other forest species with headwater designs can consolidate biodiversity management areas.

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