IRST Amphibian Literature Review – AMPC Questions 1-2

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Introduction

This literature review addresses five amphibian species covered under the Oregon Private Forest Accord (PFA) Aquatic Habitat Conservation Plan (HCP): Cope's giant salamander (*Dicamptodon copei*), coastal giant salamander (*D. tenebrosus*), Columbia torrent salamander (*Rhyacotriton kezeri*), southern torrent salamander (*R. variegatus*), and coastal tailed frog (*Ascaphus truei*). The review synthesizes available scientific information relevant to the amphibian research questions developed by the Adaptive Management Program Committee (AMPC) in November 2025. This review addresses AMPC Questions 1 and 2 related to amphibian distribution, population diversity, and population trends; the full set of AMPC questions and the scope of this review are provided in Appendix A.

These five species share many life-history characteristics that are relevant to interpreting their distribution, habitat associations, and population connectivity. All are stream-associated amphibians with aquatic and semi-aquatic life stages and biphasic life cycles. These species reproduce and lay eggs in cold, perennial streams or stream-adjacent habitats, and larval development typically takes place in flowing water over extended periods. Adults are closely associated with moist microhabitats within or near headwater stream networks and exhibit limited movement relative to many terrestrial amphibians. Collectively, these traits result in strong associations with local hydrologic conditions, stream temperature regimes, and fine-scale habitat structure, and they provide important context for evaluating patterns of occurrence and population structure across landscapes (Nussbaum et al. 1983; Good and Wake 1992; Hayes and Quinn 2015).

For the purposes of this review, we focus on research relevant to the following amphibian topic areas: (1) species range and watershed-scale distribution patterns; (2) population structure, genetics, and connectivity; (3) influence of local habitat factors on fine-scale distribution; (4) available information on population status and trends for Columbia and southern torrent salamanders on private forest lands in Oregon; and (5) contextual information relevant to evaluating how forest management may influence habitat conditions and, where data allow, population persistence of stream-associated amphibians on private forest lands. The review emphasizes synthesis of existing evidence within these topical areas rather than comprehensive treatment of all aspects of amphibian ecology.

Interpretation of distributional, population diversity, and habitat association patterns for headwater stream-associated amphibians is inherently constrained by recurring limitations in data coverage, study design, spatial scale, and analytical approach (Kroll 2009). Although broad geographic range boundaries for most taxa are well established, distribution at watershed and sub-watershed scales remains incompletely characterized due to uneven survey effort and limited standardized sampling, complicating inference about fine-scale occupancy patterns. In addition, most population diversity and habitat studies on these species have relied on short-term, reach-scale surveys that use relative abundance or occupancy metrics often without correcting for detection probability, which varies among species, life stages, habitats, and sampling conditions (Kroll 2009). When detection probability is not estimated, parameter estimates may be biased, leading to potentially unreliable inference about habitat associations or population trends (Kroll 2009; MacKenzie et al. 2017). Accordingly, the information synthesized in this review is best interpreted as describing relative patterns of occurrence and habitat association, with substantial uncertainty remaining around demographic rates, population size, and persistence.

Sources considered include verified species occurrence records compiled from multiple biodiversity databases and agency data systems, peer-reviewed and gray literature relevant to Oregon and the broader Pacific Northwest, and agency-prepared summaries that synthesize existing information on species distribution, habitat associations, and management considerations. Information is organized by topical area with species-specific subsections, following a consistent structure to facilitate comparison across taxa. Throughout the review, we identify areas where available data are limited or where key questions related to distribution, connectivity, or population status have not been evaluated.

Table 1. Glossary of select biological terms used in this review.

Term	Definition
Allelic richness	A measure of genetic diversity based on the number of different versions of a gene are present in a population that accounts for sample size.
Biphasic Life Cycle	A life-history pattern in which an organism has two distinct life stages, typically an aquatic larval stage and a terrestrial or semi-terrestrial adult stage, separated by metamorphosis.
Clade	A group of organisms that includes a common ancestor and all of its descendants, representing a single evolutionary lineage.
Congener	An organism that belongs to the same taxonomic genus as another.
Geographic Distribution	The spatial pattern of a species' occurrence within its geographic range, describing where the species is present at finer spatial scales. Distribution may be patchy and is influenced by habitat, environmental conditions.
Geographic Range	The broad spatial extent over which a species is known or inferred to occur, defined by the outer limits of confirmed or historical records. Range does not imply continuous occupancy within those boundaries.
Haplotype	A specific genetic sequence or set of genetic variants inherited together, often referring to a unique DNA sequence within a defined region of the genome.
Heterozygosity	A measure of how often individuals in a population have two different versions of the same gene, which helps indicate how much genetic variety exists within the population.
Interstitial	A term describing the small spaces between particles or structures, such as the tiny gaps between rocks or gravel in a streambed, where water, air, or small organisms can move or live.
Metamorph	An individual that has recently completed metamorphosis and transitioned from the larval stage to a terrestrial or semi-terrestrial life stage.
Mitochondrial DNA	Genetic material located in the mitochondria, inherited maternally in most animals, and widely used in population genetic and phylogeographic analyses.
Metamorphosis	A developmental process involving substantial morphological, physiological, and ecological changes that transform an aquatic larva into a terrestrial or semi-terrestrial juvenile or adult.
Neoteny	A form of paedomorphosis in which development is delayed relative to reproductive development, resulting in adults that retain juvenile or larval traits.
Oviposition	The process of laying eggs.
Paedomorph	An individual that reaches sexual maturity while retaining larval morphological traits and remaining aquatic, rather than undergoing full metamorphosis.
Paedomorphosis	A developmental phenomenon in which sexually mature individuals retain larval characteristics into adulthood; in salamanders, this may occur permanently or facultatively.

Distribution

This section summarizes the known geographic distribution of the five focal amphibian species based on documented occurrences, with emphasis on patterns relevant at watershed scales. For all five focal species, broad geographic range boundaries are reasonably well established for statewide planning purposes; however, distribution at watershed (5th-field HUC) and sub-watershed scales remains incompletely characterized, limiting inference about fine-scale occupancy patterns and management effects.

Across taxa, distributional patterns span a gradient of geographic range size and ecological specialization. *Dicamptodon tenebrosus* and *Ascaphus truei* are the most broadly distributed species, occurring across multiple ecoregions and elevational zones. *Dicamptodon copei* has a more limited and disjunct distribution, confined primarily to steep-gradient headwater streams in portions of the Coast and Cascade ranges. Torrent salamanders (*Rhyacotriton* kezeri and *R. variegatus*) are highly ecologically specialized, with occurrence records often appearing patchy and largely restricted to cold, perennial headwater environments within limited portions of western Oregon. While broad distributional patterns are well documented at county and ecoregional scales, resolution at the watershed scale remains limited for all species.

Distribution summaries draw on multiple data sources that vary in spatial resolution, temporal coverage, and survey effort. In Oregon, the primary source of vetted amphibian occurrence information is the Oregon Biodiversity Information Center (ORBIC), which maintains Sensitive Species records within the NatureServe Biotics database. These records integrate verified Element Occurrences and observation data contributed by partner agencies, researchers, and conservation programs, following standardized NatureServe and Heritage Program data quality and review procedures (Faber-Langendoen et al. 2012). ORBIC summarizes documented occurrences using Element Occurrences (EOs), which represent locations or areas of known or inferred presence and are mapped at spatial resolutions appropriate to the species and data sensitivity, rather than as point locations (Faber-Langendoen et al. 2012; ORBIC 2025).

Additional sources of distributional data include county-level summaries from the USGS Amphibian Research and Monitoring Initiative (ARMI) National Amphibian Atlas, historical museum and institutional records, agency project datasets, published locality reports, and public biodiversity aggregators (e.g., iNaturalist, GBIF, USGS BISON). Records from these sources vary in spatial precision and verification status and in many cases are already incorporated into ORBIC's database following review and quality control. No single data source provides complete or systematic coverage across all watersheds, and documented occurrence patterns reflect both true distribution and variation in survey effort. Limitations in the available distribution information, including uneven spatial coverage, variable survey effort, and data gaps at watershed scales, are noted where relevant.

Species-specific distributional information is described in detail below for each focal species. Generalized range maps from Jones et al. (2005) are included (Figures 1–5) for reference to illustrate broad geographic extent and should not be interpreted as evidence of continuous occupancy or fine-scale distribution. To aid comparison across taxa and clarify the scale and strength of available evidence, Table 2 summarizes the degree to which geographic range boundaries and watershed-scale distributions are defined based on available occurrence data and the literature, along with the primary limitations affecting inference at watershed scales.

Table 2. Summary of available evidence describing geographic range boundaries and watershed-scale distribution for the five focal species. The table highlights variation among taxa in the degree to which distribution is resolved at watershed (5th-field HUC) and sub-watershed scales and identifies limitations associated with uneven occurrence data and survey effort.

Species	Range Boundary Confidence	Watershed Distribution Confidence	Primary Limitation
Cope's giant salamander (*Dicamptodon copei*)	High	Limited	Disjunct geographic range and uneven survey effort; limited standardized sampling constrains inference regarding continuity among adjacent watersheds.
coastal giant salamander (*Dicamptodon tenebrosus*)	High	Limited	Broad geographic range but distribution documented largely through nonstandardized surveys; limited standardized sampling constrains watershed-scale inference.
Columbia torrent salamander (*Rhyacotriton kezeri*)	High	Moderate	Geographic distribution confirmed by recent rangewide standardized surveys; limited standardized sampling constrains stream-network-level inference within watersheds.
southern torrent salamander (*Rhyacotriton variegatus*)	High	Limited	Broad geographic range with uneven occurrence records; limited standardized sampling constrains watershed-scale distributional inference.
coastal tailed frog (*Ascaphus truei*)	Moderate – High	Limited	Large geographic range and regional survey bias, particularly east of the Cascades; limited watershed-scale standardized sampling constrains inference

Recent published studies illustrate both the potential and the limitations of existing approaches for resolving distribution at watershed scales. Rangewide survey and modeling efforts for torrent salamanders (Thurman et al. 2025) demonstrate how standardized field sampling informed by habitat associations can be implemented across multiple watersheds to evaluate occurrence patterns at spatial scales nested within 5th-field and finer HUCs. Although such efforts were not designed to exhaustively sample individual watersheds or estimate detection-corrected occupancy, they provide a transferable framework for watershed-aware survey design and highlight where distributional uncertainty remains. In contrast, unusually intensive, long-term inventories conducted at local scales (e.g., the recent Columbia County herpetological surveys; Hakim et al. 2025) show that distribution can be resolved at 5th-field HUC and finer scales when sustained, standardized sampling is applied across a defined landscape. These studies help demonstrate that while broad range boundaries are generally well established for all focal species, comprehensive characterization of watershed-scale and sub-watershed distribution remains limited, and would require targeted, systematic survey effort.

More recently, an ongoing, multi-year effort in Oregon's northern Coast Range (Garcia et al. 2025) illustrates how watershed-scale distribution can be resolved when standardized sampling is applied broadly and explicitly accounts for imperfect detection. This effort is cited here to illustrate how fine-scale survey design and analytical advances are currently being implemented rather than to reinterpret their bearing on our understanding of species distributions at this time. Garcia et al. are conducting coordinated visual encounter and environmental DNA surveys across approximately 80 HUC-12 watersheds, incorporating repeat sampling to estimate detection-corrected occupancy for torrent salamanders, giant salamanders, and coastal tailed frogs. Preliminary results from the 2024–2025 field seasons indicate high detection probabilities for all three taxa and suggest spatial and temporal variation in occupancy associated with watershed geography (e.g., east–west drainage orientation) and stream conditions (Garcia et al. 2025). However, because these data are still under analysis and not yet publicly available, they are not currently synthesized here. When completed, this work may represent one of the most comprehensive, watershed-aware efforts underway and is likely to substantially improve inference about distribution and occupancy patterns of the species covered under the HCP within and among Oregon watersheds.

Cope's giant salamander (*Dicamptodon copei*)

Figure 1. Generalized geographic range of Cope's giant salamander (*Dicamptodon copei*) in the Pacific Northwest, showing the species' known range within Oregon and adjacent regions. Map reproduced with permission from Jones et al. (2005).

Geographic range and distribution: *D. copei* is endemic to the Pacific Northwest and occurs in two geographically disjunct regions within the Coast and Cascade ranges of Oregon and Washington (Foster and Olson 2014). The species' range extends from the northwestern Olympic Peninsula south to the Nehalem River watershed in the Oregon Coast Range, and from the Nisqually River at Mount Rainier south through the Cascade Range to the upper White River watershed in Wasco County, Oregon (Foster and Olson 2014, Foster et al. 2015, Bury et al. 2014, 2017).

In Oregon, *D. copei* is documented from portions of the northern Coast Range and Cascades, including Clackamas, Clatsop, Columbia, Hood River, Multnomah, Tillamook, Washington, and Wasco counties (Nussbaum 1970; Foster and Olson 2014; Foster et al. 2015; Hakim et al. 2025; ORBIC 2025; USGS 2025). Occurrences east of the Cascade Crest have been confirmed in tributaries of the lower White River watershed, extending the known distribution up to approximately 50 km (~31 miles) east of the crest (Bury et al. 2014, 2017). The species overlaps with its congener *D. tenebrosus* in portions of the northern Coast Range and Cascades. Spatially, *D. copei* exhibits a patchy and discontinuous distribution, with known occurrences documented from 582 sites distributed across 165 sixth-field (HUC12) watersheds, reflecting an uneven distribution among headwater networks (Foster et al. 2015).

Elevational range: Documented occurrences of *D. copei* span elevations from sea level in Washington to about 1600 m (~5250 ft) in the East Cascades ecoregion of Oregon, with a reported mean elevation of approximately 475 m (1558 ft) (Foster and Olson 2014).

Distribution data gaps: Although the broad geographic range of *D. copei* is reasonably well established, documented occurrences are unevenly distributed across watersheds and reflect variable survey effort rather than systematic coverage. Standardized sampling at the 5th-field HUC scale has been limited, constraining inference regarding fine-scale occupancy patterns and continuity among adjacent watersheds, particularly within the Coast and Cascade ranges. In addition, the predominance of opportunistic and non-standardized records limits assessment of temporal changes in distribution. As a result, watershed-scale distribution patterns and potential gaps between known localities remain incompletely resolved.

coastal giant salamander (*Dicamptodon tenebrosus*)

Geographic range and distribution: The range of *D. tenebrosus* extends from British Columbia south into California and is widely distributed in western Oregon across the Coast Range, western Cascades, and Klamath Mountains (Nussbaum 1976; Good and Wake 1992; Olson and Weaver 2007). In Oregon, the species is documented from near-coastal watersheds inland to the Cascades and into the Klamath region, with isolated records occurring east of the Cascade crest in the Deschutes River basin (Bury et al. 2017). County records include Benton, Clackamas, Clatsop, Columbia, Coos, Curry, Douglas, Hood River, Jackson, Josephine, Lane, Lincoln, Linn, Marion, Multnomah, Polk, Tillamook, Wasco, Washington, and Yamhill counties (ORBIC 2025, USGS 2025). The species overlaps with *D. copei* in portions of the northern Coast Range and western Cascades in Washington but extends substantially farther south into southwestern Oregon and northern California.

Figure 2. Generalized geographic range of coastal giant salamander (*Dicamptodon tenebrosus*) in the Pacific Northwest, showing the species' known range within Oregon and adjacent regions. Map reproduced with permission from Jones et al. (2005).

Elevational range: Documented occurrences span from near sea level to approximately 2160 m (7000 ft) elevation (AmphibiaWeb 2025).

Distribution data gaps: While the broad geographic range of *D. tenebrosus* in western Oregon is well established, documented occurrences are unevenly distributed across watersheds and reflect variable survey effort rather than systematic coverage. Standardized sampling at the 5th-field HUC scale has been limited, constraining inference regarding fine-scale occupancy patterns and continuity among adjacent watersheds. In addition, the predominance of opportunistic and non-standardized records limits assessment of temporal changes in distribution. Consequently, watershed-scale distribution patterns remain incompletely resolved.

Columbia torrent salamander (*Rhyacotriton kezeri*)

Figure 3. Generalized geographic range of Columbia torrent salamander (*Rhyacotriton kezeri*) in the Pacific Northwest, showing the species' known range within Oregon and adjacent regions. Map reproduced with permission from Jones et al. (2005).

Geographic range and distribution: *R. kezeri* is restricted to coastal and near-coastal regions of northwestern Oregon and southwestern Washington, extending from the Little Nestucca River system in Tillamook County, Oregon, north to the Chehalis River in Grays Harbor County, Washington (Good and Wake 1992). In Oregon, the species is confined to the Coast Range ecoregion, with confirmed occurrences documented from Clatsop, Columbia, Tillamook, Washington, Yamhill, and Polk counties (Good and Wake 1992; ORBIC 2025, USGS 2025). Within this geographic range, distribution is non-uniform and structured across multiple spatial scales: populations are widespread but patchily distributed among watersheds (Russell et al. 2004).

Elevational range: Across its range, *R. kezeri* has been documented from near sea level to approximately 1000 m (3000 ft) elevation. The highest reported occurrences include Boisfort Peak at 948 m (3110 ft) in the Willapa Hills of southwestern Washington and Saddleback Mountain at 1001 m (3284 ft) in the Coast Range of northwestern Oregon (Good and Wake 1992; AmphibiaWeb 2025).

Distribution data gaps: The broad geographic range of *R. kezeri* is well defined and recent standardized headwater surveys confirm occurrence across much of the species' recognized range (Thurman et al. 2025), however, documented occurrences remain unevenly distributed across watersheds and do not reflect comprehensive, standardized coverage at the 5th-field HUC scale. Distribution within individual watersheds and across connected headwater stream networks remains less well characterized, constraining inference regarding fine-scale occupancy patterns and continuity among adjacent watersheds. In addition, the predominance of historical, nonstandardized records limits assessment of temporal changes in distribution.

southern torrent salamander (*Rhyacotriton variegatus*)

Geographic range and distribution: *R. variegatus* occurs in the Coast Range from approximately Mendocino County, California, north to the Little Nestucca River in Oregon (Good and Wake 1992). The species has a broad range but is patchily distributed in western Oregon across the Coast Range, Klamath Mountains, western Cascades, and marginally within portions of the Willamette Valley ecoregion (Good and Wake 1992; Wagner et al. 2006; Olson and Ares 2022). Verified occurrence records document the species in Benton, Coos, Curry, Douglas, Josephine, Lane, Lincoln, Polk, Tillamook, and Yamhill counties (ORBIC 2025, USGS 2025).

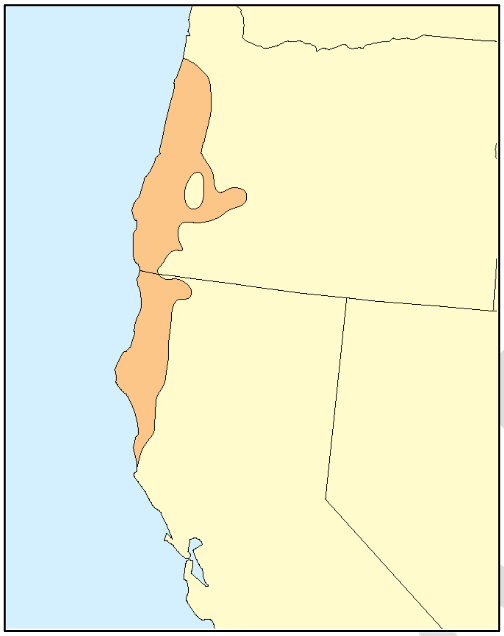

Figure 4. Generalized geographic range of southern torrent salamander (*Rhyacotriton variegatus*) in the Pacific Northwest, showing the species' known range within Oregon and adjacent regions. Map reproduced with permission from Jones et al. (2005).

A geographically disjunct population occurs in the western Cascades of Douglas and Lane counties, Oregon, between the Willamette and Umpqua river basins (Good and Wake 1992; Wagner et al. 2006). In this region, *R. variegatus* abuts populations of *R. cascadae*; mitochondrial DNA indicate no sympatry or hybridization, and the Middle Fork Willamette River appears to function as a barrier between the species (Wagner et al. 2006).

Elevational range: Documented occurrences of *R. variegatus* span from sea level to approximately 1469 m (4820 ft) elevation, with most records associated with cool, montane headwater environments (Welsh and Lind 1996).

Distribution data gaps: Although broad range boundaries for *R. variegatus* are well established, documented occurrences are unevenly distributed across watersheds and reflect variable survey effort rather than systematic coverage. Standardized sampling at the 5th-field HUC scale has been limited, constraining inference regarding fine-scale occupancy patterns and continuity among adjacent watersheds. In addition, the predominance of opportunistic and non-standardized records limits assessment of temporal changes in distribution. As a result, watershed-scale distribution patterns remain incompletely resolved.

coastal tailed frog (*Ascaphus truei*)

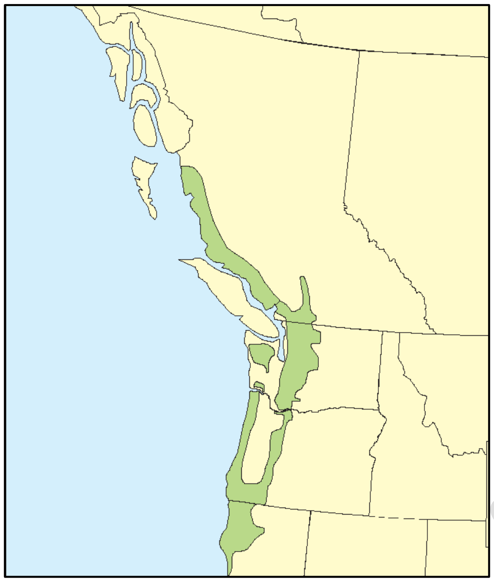

Figure 5. Generalized geographic range of coastal tailed-frog (*Ascaphus truei*) in the Pacific Northwest, showing the species' known range within Oregon and adjacent regions. Map reproduced with permission from Jones et al. (2005).

Geographic range and distribution: *A. truei* ranges from southern British Columbia to northwestern California (Nussbaum et al. 1983; Hayes and Quinn 2015). In Oregon, the species occurs in the Coast Range, West Cascades, East Cascades, and Klamath Mountains ecoregions (Hayes and Quinn 2015; ORBIC 2025). Occurrence records document the species in Benton, Clackamas, Clatsop, Columbia, Coos, Curry, Deschutes, Douglas, Hood River, Jackson, Jefferson, Josephine, Klamath, Lane, Lincoln, Linn, Marion, Multnomah, Polk, Tillamook, Wasco, Washington, and Yamhill counties (ORBIC 2025, USGS 2025). Most occurrences are concentrated west of the Cascade crest; however, a small number of records have been documented east of the Cascade Mountains, including near low-elevation mountain passes (Hayes and Quinn 2015).

At broad spatial scales, watershed-level occupancy appears high in some regions but variable among landscapes. In surveys of randomly selected third-order watersheds in Oregon and Washington (i.e., fine-scale sub-watershed sampling units substantially smaller than 5th-field HUCs), coastal tailed frogs were detected in approximately 73% of watersheds (22 of 30) in the Oregon Coast Range and 80% of watersheds (8 of 10) in the southern Cascades, but only 30% of watersheds (8 of 27) in the Willapa Hills of southwestern Washington, indicating pronounced regional variation in distribution (Kroll et al. 2010). Repeated surveys of individual streams further demonstrate that detection probability is imperfect and that single-visit surveys substantially underestimate presence and local abundance, particularly later in the season (Kroll et al. 2008).

Elevational range: Documented occurrences of *A. truei* span from sea level to approximately 2074 m (6804 ft) elevation (Washington Herp Atlas 2017).

Distribution data gaps: The broad geographic range of *A. truei* is well characterized across the Pacific Northwest, though the exact boundaries of its range margin east of the Cascades could be improved by additional focused surveys. Documented occurrences are unevenly distributed across watersheds and reflect variable survey effort rather than systematic coverage. Standardized sampling at the 5th-field HUC scale has been limited, constraining inference regarding fine-scale occupancy patterns and continuity among adjacent watersheds. In addition, the predominance of opportunistic and non-standardized records limits assessment of temporal changes in distribution. As a result, watershed-scale distribution patterns remain incompletely resolved.

Population Diversity

Population diversity in stream-associated amphibians encompasses variation in local abundance and occupancy, demographic and life-history characteristics, age and life-stage structure, spatial distribution across stream networks, and genetic differentiation among populations. Together, these dimensions may influence population persistence in headwater stream environments. However, for many stream-associated amphibians, direct measurement of these attributes is constrained by cryptic behavior, prolonged aquatic life stages, reliance on interstitial refugia, and the logistical challenges of long-term monitoring in headwater systems.

This section synthesizes available evidence on population diversity for the five focal species, drawing from site-scale abundance and occupancy studies, demographic and life-history investigations, and population genetic analyses conducted in Oregon and elsewhere within each species' range. Rather than estimating population size or demographic exchange directly, which remains infeasible for most taxa, this section evaluates how well different components of population diversity have been characterized and identifies recurring patterns of spatial heterogeneity and uncertainty. Accordingly, the population diversity evidence summarized below primarily reflects relative patterns and qualitative differences, rather than direct estimates of population size or demographic rates.

Across species, local abundance is most commonly described using relative density or occupancy metrics at individual stream or reach scales. These studies consistently document substantial spatial variation within and among watersheds but do not provide rangewide or watershed-scale abundance estimates. Demographic information is uneven and frequently biased toward early aquatic life stages, with limited direct data on survival, recruitment, or population turnover for later life stages. As a result, important gaps persist for all species in standardized abundance estimation, demographic rate quantification, and assessment of the temporal stability of local populations. Genetic studies offer valuable insight into population structure and differentiation among basins and regions, but the extent and resolution of Oregon-specific genetic information vary widely among taxa.

Species-specific population diversity information is described in detail below for each focal species. To facilitate comparison across taxa and clarify the scope and strength of available evidence, Table 3 summarizes how well different components of population diversity are characterized for each species, along with key limitations affecting inference about population diversity, including spatial variation in local abundance, demographic and life-history parameters, age or life-stage structure, and population genetic structure.

Table 3. Summary of evidence characterizing population diversity for five focal species.

Species	Local abundance & spatial heterogeneity	Demography & life history	Life-stage structure	Genetic structure & connectivity
Cope's giant salamander (*Dicamptodon copei*)	Site-scale density estimates and patchy occupancy across headwater networks documented; substantial among-stream variation; local density may not reflect stable populations; no rangewide abundance estimates	Demographic rates remain poorly quantified. Available information is largely limited to larval abundance, size structure, and qualitative observations of developmental pathways, with little information on survival, recruitment, or life-stage transitions across habitats or regions	Early age–size structuring and possible age-class segregation inferred; populations dominated by larval and paedomorphic stages; age inference unreliable beyond early larval years	Strong structure among watersheds and major barriers (e.g., Columbia River); Oregon sampling limited and spatially uneven
coastal giant salamander (*Dicamptodon tenebrosus*)	Pronounced among-stream heterogeneity in larval density and patchy occupancy documented at reach and watershed scales; strong interannual variability; no rangewide abundance estimates	Age- and season-specific apparent larval survival quantified; life-history plasticity documented (metamorphosis vs. paedomorphosis); demographic rates for metamorphs and terrestrial adults largely unknown	Multiple larval age classes documented; populations numerically dominated by larvae in surveys; size-based age inference unreliable and adult stages underrepresented	Strong differentiation at broad geographic scales; Oregon-specific structure and within-state connectivity partially resolved
Columbia torrent salamander (*Rhyacotriton kezeri*)	Incomplete occupancy across headwater stream networks; substantial among-site variation in local abundance documented at reach and landscape scales; local density may not reflect stable populations; no rangewide abundance estimates	Life history with prolonged, multi-year larval development; demographic inference based primarily on life-stage composition and abundance patterns; survival and recruitment rates unquantified	Populations numerically dominated by larval and juvenile stages; life-stage composition inferred from captures; adult stages infrequently detected	Strong constraints on connectivity imposed by large streams, land cover, and road density; relatively low within-population diversity; Oregon sampling limited and spatially uneven
southern torrent salamander (*Rhyacotriton variegatus*)	Patchy occupancy and wide variation in local density across landscapes; abundance strongly dependent on microhabitat targeting and sampling design; local density may not reflect stable populations; no rangewide abundance estimates	Slow life history with prolonged larval development and delayed maturity; limited direct estimates of survival; demographic inference largely based on life-stage composition	– Prolonged larval period and multiple juvenile age classes documented; populations numerically dominated by larval and juvenile stages; adult abundance and survival poorly characterized	Deep regional structure with strong phylogeographic breaks associated with major rivers; high genetic differentiation across range; Oregon structure partially resolved but uneven (e.g., disjunct western Cascades population).
coastal tailed frog (*Ascaphus truei*)	Strong heterogeneity in occupancy and local abundance across watersheds; locally dense larval populations documented alongside persistent low-density populations; larval abundance may not reflect adult population size or stability; no rangewide abundance estimates	Prolonged larval development and delayed recruitment documented; demographic inference largely constrained by high mortality of early life stages and limited observation of adults; adult survival and recruitment unquantified	Populations strongly dominated by larvae; metamorphs and juveniles detected less frequently; adults rare and cryptic in surveys, limiting inference about population structure	Species-level divergence from *A. montanus* well supported; fine-scale population structure within Oregon unresolved; evidence indicates potential for terrestrial connectivity in intact forests

Cope’s giant salamander (Dicamptodon *copei*)

Local abundance and spatial heterogeneity: Although long-term population trends and most demographic parameters for *D. copei* remain poorly characterized, available evidence indicates limited but measurable variation in age–size structure, spatial distribution, and local abundance. Foster et al. (2015) report that, aside from some unpublished data, little information exists on age–size structure in *D. copei* populations; however, surveys on the Olympic Peninsula suggest some age–size segregation, with larval and paedomorphic individuals exhibiting a bi-modal size distribution corresponding to two apparent age classes in small to intermediate headwater streams (Foster et al. 2015; A.D. Foster, unpublished data). Age-class inference beyond the first two to three years is considered unreliable for *Dicamptodon* spp., and size-class modality should therefore be interpreted cautiously (Foster et al. 2015; Nussbaum and Clothier 1973).

Regional differences in abundance have been noted, with populations reaching their highest apparent densities in the Willapa Hills and South Cascades of Washington and lowest densities in the Olympic Peninsula, where populations are exclusively *D. copei* (Spear et al. 2011; Foster et al. 2015). Although population-wide abundance estimates are lacking, several studies have quantified local in-stream densities, including mean densities of approximately 0.16 individuals/m^2 in Olympic Peninsula streams (Adams and Bury 2002), 0.30 individuals/m^2 in July and 0.17 individuals/m^2 in September in Willapa Hills streams surveyed over multiple years (Foster et al. 2015) and a broader range of 0.06–1.4 individuals/m^2 in headwater streams of southern Skamania County, Washington (Steele et al. 2002). Collectively, these data indicate that while comprehensive demographic characterization is lacking, *D. copei* populations exhibit measurable spatial heterogeneity in distribution, local abundance, and limited age–size structure at site scales, though differences in field methods among studies limit direct comparison of density and related estimates.

Demographic and life-history variation: Demographic information for *D. copei* is limited, but available evidence indicates substantial life-history variation within populations. The species was originally described as an obligate paedomorph; however, fully metamorphosed terrestrial adults have since been documented, including observations of more than 50 post-metamorphic individuals in the Willapa Hills of southwestern Washington (Wagner 2015). Although the environmental and physiological triggers for metamorphosis remain poorly understood, these observations indicate that metamorphosis occurs at non-trivial frequencies in at least some populations rather than being restricted to rare individuals (Wagner 2015). Consequently, *D. copei* is now recognized as exhibiting facultative paedomorphosis, with larvae, paedomorphic adults, and metamorphosed adults co-occurring within the species (Spear et al. 2011; Wagner 2015).

Population genetics: *Dicamptodon copei* was described as a distinct species based on morphological and life-history differences, including obligate neoteny and sympatric occurrence with *D. ensatus* (Nussbaum 1970). Subsequent genetic and phylogenetic analyses confirmed this species-level separation and indicate that *D. copei* represents an ancient lineage whose divergence predates the Pleistocene (Daugherty et al. 1983; Steele et al. 2005). Available genetic studies indicate pronounced population genetic structure and limited connectivity at multiple scales in *D. copei*. At broader spatial scales, deep genetic divergence among regions and drainage systems reflects long-term isolation and historical separation among major lineages, with additional structure evident within species (Daugherty et al. 1983; Steele et al. 2005, 2007). The Columbia River functions as a major barrier to connectivity, with no shared haplotypes (i.e., no identical genetic sequences) detected between populations north and south of the river, and represents the strongest documented discontinuity in the species' genetic structure (Steele et al. 2005, 2007). At finer spatial scales, genetic differentiation is frequently partitioned among headwater catchments and stream networks, reflecting limited dispersal along streams. Genetic patterns indicate effective dispersal distances of up to approximately 5 km among genetically homogeneous localities, with limited exchange among tributaries beyond this scale (Steele et al. 2009; Spear et al. 2011). Landscape-genetic analyses indicate that while major rivers may act as barriers, stream corridors facilitate gene flow where hydrologic connectivity is maintained, but drier conditions and more fragmented forest cover are associated with restricted gene flow and increased genetic differentiation (Trumbo et al. 2013). These patterns are consistent with the species' largely aquatic life history and limited overland dispersal capacity.

Population diversity data gaps: Rangewide population size and long-term trends for *Dicamptodon copei* remain unknown, and available information is geographically uneven and largely derived from site-scale surveys rather than standardized, long-term monitoring. While multiple studies report local densities and regional differences in apparent abundance, comparable estimates across the species' range are lacking, and it remains unclear how consistently local density metrics reflect stable breeding populations versus temporally variable occupancy.

Demographic rates such as survival and life-stage transitions are poorly quantified, limiting inference about population stability and turnover. Existing evidence suggests some age–size structuring and possible age-class segregation in headwater streams; however, age inference beyond early years is unreliable in *Dicamptodon* spp., and size-class modality should be interpreted cautiously (Foster et al. 2015; Nussbaum and Clothier 1973). Genetic studies document strong population structure and major barriers to gene flow (notably the Columbia River) and indicate limited effective dispersal at local scales, but Oregon genetic sampling remains limited and concentrated in a small portion of the species' range, leaving population structure, genetic diversity, and connectivity across most of Oregon poorly characterized (Steele et al. 2005, 2007, 2009; Spear et al. 2011; Trumbo et al. 2013).

coastal giant salamander (*Dicamptodon tenebrosus*)

Local abundance and spatial heterogeneity: *D. tenebrosus* exhibits a broader and more connected geographic distribution than its congener, *D. copei*, a pattern attributed to greater ecological flexibility, more frequent terrestrial dispersal, and reduced population genetic isolation (Foster et al. 2015). Despite this broad-scale connectivity, local populations of *D. tenebrosus* are characterized by pronounced spatial and temporal heterogeneity in occupancy and abundance across headwater stream networks. For example, at the northern range margin in British Columbia, larval densities varied widely among streams and were lower in streams affected by forest harvest than in less disturbed forest contexts, indicating substantial among-site variation in local abundance (Ferguson 1998). Detections across stream networks was patchy, with larvae detected in only a subset of apparently suitable streams within a watershed, suggesting frequent local absence or extirpation (Ferguson 1998). Additional evidence for fine-scale population heterogeneity is provided by Steele et al. (2002), who surveyed first-order streams on managed timberlands in southern Skamania County, Washington, over two consecutive summers. Larval *D. tenebrosus* were detected in 16 of 29 streams (55%) in 1998 and in 16 of 33 streams (48%) in 1999, indicating patchy occupancy even among apparently suitable headwater channels. Where present, local larval densities varied widely, ranging from 0.05–0.79 individuals/m^2 in 1998 and 0.06–2.5 individuals/m^2 in 1999. Densities were significantly correlated between years ($r = 0.44$, $p = 0.02$) but exhibited high interannual variance, suggesting unstable local abundance and dynamic persistence at the stream scale (Steele et al. 2002). Recent reach-scale depletion surveys in 24 fish-bearing headwater streams in the Oregon Coast Range documented wide variation in population density (0.009–0.498 individuals/m^2) and biomass density (0.027–2.67 g/m^2), even among streams of similar size and forest condition, reinforcing pronounced local-scale heterogeneity in abundance (Neal et al. 2024).

Demographic and life history variation: Demographic information for *D. tenebrosus*, remains incomplete, but is better characterized than for more strictly headwater-restricted taxa, with evidence for strong age-structured variation during the larval stage and pronounced life-history plasticity. A two-year mark–recapture study across 14 Oregon Coast Range streams demonstrated substantially lower apparent annual survival for first-year larvae than for second- and third-year larvae, indicating strong differences in cohort persistence and recruitment success (Sagar et al. 2007). Apparent survival also varied seasonally, with higher survival during non-summer periods for both age classes, suggesting temporal variability in demographic rates within occupied streams (Sagar et al. 2007); however, because apparent survival reflects both true survival and availability for capture, these patterns may partially reflect larval movement outside sampled stream reaches or into subsurface or off-channel habitats during low-flow periods rather than mortality alone.

Background studies further indicate that *D. tenebrosus* exhibits alternative developmental pathways, with larvae either metamorphosing into terrestrial adults or remaining as paedomorphic aquatic adults following a larval period typically lasting two to three years in Oregon (Nussbaum and Clothier 1973). Larval stage length appears to vary geographically, with northern populations exhibiting slower growth and prolonged larval periods, indicating potential for regional variation in age structure and recruitment dynamics (Ferguson 1998). Population structure varies predictably with aquatic habitat type, with regularly intermittent streams unable to support persistent larval populations and permanent streams differing in the number of larval age classes present and the frequency of paedomorphosis (Nussbaum and Clothier 1973).

Recent reach-scale surveys in the Oregon Coast Range confirm that stream populations are numerically dominated by small larvae, with relatively few large larvae or aquatic adults detected, reinforcing a strong life-stage bias in available demographic information (Neal et al. 2024). While aquatic adults were observed infrequently in the Sagar et al. (2007) study, their presence is consistent with this broader life-history framework. In larger permanent streams, ponds, and lakes, however, paedomorphic adults may constitute a substantial portion of the breeding population, whereas metamorphosis is more common and neoteny rare in smaller streams (Nussbaum and Clothier 1973).

Population genetics: *Dicamptodon tenebrosus* was distinguished from the historically broad *D. ensatus* species concept based on consistent morphological differentiation between northern coastal populations and the more southern California lineage of Pacific giant salamander (*D. ensatus* sensu stricto) (Nussbaum 1976). Rangewide genetic studies indicate substantial population differentiation in *D. tenebrosus* across broad geographic and ecoregional scales (Good and Wake 1992; Auteri et al. 2022). Analyses of mitochondrial DNA identify major regional lineages, broadly corresponding to coastal and interior regions and reflecting long-term historical isolation (Good and Wake 1992). Oregon populations occur near the center of the species' range and span multiple physiographic regions, including the Coast Range, Cascades, and Klamath Mountains. Although this geographic position suggests potential contact among divergent lineages, Oregon-specific phylogeographic resolution remains limited. Recent genetic analyses further indicate that watershed boundaries do not strictly delimit population structure at local scales, suggesting gene flow may occur among nearby headwater drainages (Auteri et al. 2022). Genetic evidence suggests that terrestrial movement in this species may contribute to gene flow among nearby headwater drainages where aquatic connectivity is absent (Auteri et al. 2022), a pattern consistent with broader conceptual models emphasizing the importance of terrestrial connectivity among headwater stream–riparian habitats (Olson et al. 2007).

At finer spatial scales, forest management context may influence genetic diversity and population structure in *D. tenebrosus*. In British Columbia, a small study of *D. tenebrosus* in managed forests compared to old growth forest suggested that allelic richness and heterozygosity were lower in 3–9-year-old clearcuts relative to second-growth and old-growth forests, consistent with population bottlenecks associated with harvest disturbance (Curtis and Taylor 2003). Although genetic differentiation among forested sites was modest, clearcut populations were more genetically distinct from reference old-growth sites, and recovery of genetic diversity was inferred to occur over centuries rather than decades (Curtis and Taylor 2003). However, the authors acknowledge uncertainty in the potential consequences of variation in genetic diversity and that *D. tenebrosus* has moderate dispersal ability, which can enhance gene flow among populations. These findings suggest that while local gene flow can occur among headwater populations, forest disturbance may impose long-lasting genetic effects that are not captured by short-term demographic responses.

Population diversity data gaps: Rangewide population size and long-term trends for *D. tenebrosus* remain unknown. Available evidence is derived primarily from site-scale occupancy, density, and short-term demographic studies rather than standardized, long-term monitoring, limiting inference about population stability and persistence. Local larval abundance and occupancy vary widely among streams and among years, and it remains unclear how consistently observed density patterns reflect stable breeding populations versus dynamic local extinction–recolonization processes (Ferguson 1998; Steele et al. 2002).

Demographic parameters remain incompletely quantified and strongly biased toward larval stages. While mark–recapture data demonstrate pronounced age-specific and seasonal variation in apparent larval survival, comparable estimates for older larvae, metamorphs, terrestrial adults, and paedomorphic adults are largely unavailable, limiting inference about recruitment, turnover, and lifetime reproductive contribution (Sagar et al. 2007). The relative frequency and spatial distribution of alternative developmental pathways (metamorphosis versus paedomorphosis) likely vary with habitat type and hydroperiod, but these strategies have not been quantified systematically across Oregon or across the species' range (Nussbaum and Clothier 1973; Ferguson 1998).

Genetic studies document substantial population differentiation at broad geographic scales and suggest some gene flow among nearby headwater drainages, but Oregon-specific phylogeographic resolution and within-state population structure remain limited relative to management needs (Good and Wake 1992; Auteri et al. 2022).

Columbia torrent salamander (*Rhyacotriton kezeri*)

Local abundance and spatial heterogeneity: Available information indicates that *R. kezeri* populations are spatially uneven and discontinuously distributed across headwater stream networks, though overall population size and long-term trends remain unknown. Surveys in managed forests of Oregon's Coast Range detected the species in approximately 58% of sampled headwater streams, indicating incomplete occupancy even within broadly suitable habitat (Russell et al. 2004). Similar patterns have been reported in Washington, where *R. kezeri* occurred in 53% of surveyed perennial headwater streams in 50–65-year-old managed stands, further supporting patchy distribution across headwater networks (Wilkins and Peterson 2000). Within occupied streams, abundance varies markedly among sites. In the Oregon Coast Range, relative salamander abundance at the landscape scale ranged from 0 to 4.77 individuals per linear meter of stream (mean = 0.47 salamanders/m), demonstrating pronounced among-site variability in local population size (Russell et al. 2004). At the stream-reach scale, mean density was estimated at 2.24 ± 0.89 salamanders/m^2, indicating substantial fine-scale heterogeneity in local abundance even within occupied watersheds (Russell et al. 2004).

Demographic and life history variation: Demographic information for *R. kezeri* is limited, with most inference drawn from patterns of abundance and life-history stage composition rather than direct estimates of survival or recruitment. Available evidence indicates a slow life history characterized by multi-year larval development, often lasting two to four years (Nussbaum and Tait 1977; Nussbaum et al. 1983; Russell et al. 2004). Life-stage composition within occupied streams is strongly skewed toward aquatic juveniles. Larvae comprised approximately 60–63% of captures across both landscape and reach scales, indicating that populations are typically numerically dominated by larval and juvenile stages (Russell et al. 2004), likely reflecting a combination of true demographic structure associated with small clutch sizes and multi-year larval development, and perhaps detection bias favoring aquatic larvae over more cryptic or less frequently sampled adults.

Population genetics: Genetic evidence indicates that population connectivity in *R. kezeri* is strongly constrained by landscape features. Large streams and rivers function as barriers to gene flow; notably, no genetic exchange was detected between *R. kezeri* and *R. variegatus* where their ranges approach near the Little Nestucca River, despite close geographic proximity (Good and Wake 1992). Population genetic analyses further indicate relatively low genetic diversity within *R. kezeri* populations, likely reflecting the species' restricted distribution and limited connectivity among headwater watersheds (Emel et al. 2019). Landscape-genetic analyses demonstrate that land cover and road density significantly influence genetic structure, with forested habitats facilitating connectivity and developed or non-forested areas associated with increased genetic differentiation (Emel et al. 2019). Fragmentation effects vary spatially across the species' range, reflecting heterogeneity in forest cover and road density (Emel et al. 2019).

Population diversity data gaps: Rangewide population size and long-term trends for *R. kezeri* remain unknown. Available evidence is derived primarily from site-scale occupancy and relative abundance surveys conducted over limited time frames, rather than standardized, long-term monitoring, constraining inference about population stability and persistence. Occupancy is incomplete across headwater stream networks, and local abundance varies widely among occupied sites; however, it remains unclear whether occupied channels represent stable demographic units or whether occupancy and density fluctuate substantially over longer temporal scales (Russell et al. 2004; Wilkins and Peterson 2000).

Demographic parameters are incompletely quantified and are inferred primarily from life-stage composition rather than direct estimates of survival or recruitment. Available evidence indicates prolonged larval development, often lasting multiple years, and populations numerically dominated by larval and juvenile stages; however, adult abundance and survival are poorly characterized, limiting assessment of population diversity.

Genetic studies indicate relatively low within-population diversity, strong constraints on connectivity imposed by rivers, land cover, and road density, and sensitivity of population structure to landscape fragmentation. However, genetic sampling remains spatially limited across Oregon and across the species' full range, restricting inference about watershed-scale population structure and the generality of observed fragmentation effects (Good and Wake 1992; Emel et al. 2019).

southern torrent salamander (*Rhyacotriton variegatus*)

Local abundance and spatial heterogeneity: Available information indicates that *R. variegatus* populations are spatially heterogeneous across their range, with occupancy strongly dependent on local habitat conditions, though overall population size and trends remain unknown. In northern California, the species was detected in 80.3% of surveyed headwater streams in managed forest landscapes (Diller and Wallace 1996). In contrast, occupancy in the Klamath Mountains of northern California was substantially lower when sites were selected randomly (28.2–36.8%), but increased to 46.6–62.3% at sites with suitable microhabitat characteristics, indicating strong patchiness and habitat filtering at the landscape scale (Welsh and Lind 1992).

Local abundance and density vary widely among occupied sites. In Lincoln County, Oregon, Nussbaum and Tait (1977) documented 216 larvae within a 16.5 m^2 area, corresponding to a mean density of 12.9 larvae/m^2 (95% CI: 8.7–17.2). Across California, densities in suitable habitat ranged from 1 to 50 individuals per 10 m^2 plot, with a mean of 0.68 individuals/m^2 (SD = 0.89) (Welsh and Lind 1996). In managed forests of northern California, salamander densities ranged from 0.014 to 5.0 individuals/m^2 depending on sampling scale and habitat type, with mean densities of approximately 0.12 individuals/m^2 in randomly selected stream reaches and approximately 0.28 individuals/m^2 in targeted microhabitat surveys, illustrating pronounced among-site variability in local population size (Diller and Wallace 1996).

Demographic and life history variation: Demographic information suggests a slow life history characterized by prolonged development and delayed maturity. Southern torrent salamanders typically require approximately 3–3.5 years to complete larval development, followed by an additional 1–1.5 years to reach sexual maturity, and egg incubation alone may last up to approximately 290 days, resulting in a generation time approaching five years in some individuals (Nussbaum and Tait 1977; Nussbaum et al. 1983; Tait and Diller 2006). Available demographic data from California indicate an approximately even sex ratio and populations numerically dominated by juveniles, with juveniles comprising approximately 80% of captures (Welsh and Lind 1992). Estimated annual survival for *R. variegatus* in California was 0.44 (SE = 0.76), consistent with slow turnover and strong age structure (Welsh and Lind 1992).

Population genetics: Genetic studies indicate pronounced population structure in *R. variegatus* across its geographic range, with strong differentiation among regions consistent with long-term isolation in headwater environments. Phylogenetic analyses identify three major evolutionary lineages corresponding to northern Oregon, central Oregon, and northern California, reflecting deep lineage separation shaped by historical river barriers and climatic processes (Good and Wake 1992; Wagner 2000; Miller et al. 2006). Major river systems, including the Yaquina River in Oregon and the Smith River in California, act as phylogeographic breaks between clades and serve as barriers to gene flow (Miller et al. 2006).

A geographically disjunct population occurs in the western Cascades between the Willamette and Umpqua Rivers. Large streams and rivers, including the Yaquina, Smith, and Willamette Rivers, function as barriers to connectivity among clades (Good and Wake 1992; Miller et al. 2006; Wagner et al. 2006). Landscape-genetic analyses confirm high levels of genetic structure across the species' range, with populations in the northern portion of the range exhibiting greater fragmentation than those to the south (Emel and Storfer 2015; Emel et al. 2019). Forested land cover and intact forest canopy are associated with greater genetic connectivity, while developed or non-forested land cover and road density are associated with increased genetic differentiation (Emel and Storfer 2015; Emel et al. 2019). Field observations indicate that most individuals are detected within a few meters of flowing water, with rare observations up to approximately 160 m (525 ft) from permanent streams, suggesting limited terrestrial movement away from headwater channels under wet conditions (Welsh and Lind 1992; Leppin et al. 2020). Although *R. variegatus* populations are known to persist following forest disturbance (e.g., Kroll et al. 2008; Olson and Ares 2022), the mechanisms by which connectivity is maintained or re-established after disturbance remain poorly understood.

Population diversity data gaps: Rangewide population size and long-term trends for *R. variegatus* remain unknown. Available evidence is derived primarily from site-scale occupancy and density surveys that vary in sampling design and microhabitat targeting, limiting inference about baseline conditions, temporal stability, and comparability among regions. Occupancy and local abundance are strongly influenced by microhabitat suitability, and reported densities span a wide range; however, it remains unclear how consistently these metrics reflect stable breeding populations versus localized concentrations detected under favorable sampling conditions (Welsh and Lind 1992, 1996; Diller and Wallace 1996; Nussbaum and Tait 1977).

Demographic parameters are incompletely quantified and are inferred primarily from life-stage composition rather than direct estimates of survival or recruitment. Available evidence indicates prolonged larval development and populations numerically dominated by larval and juvenile stages, but adult abundance and survival are poorly characterized, limiting assessment of population diversity (Welsh and Lind 1992; Nussbaum and Tait 1977; Tait and Diller 2006).

Genetic studies document deep regional structure, strong phylogeographic breaks associated with major rivers, and spatially variable fragmentation effects across the species' range. However, Oregon-specific resolution remains uneven, particularly for the geographically disjunct western Cascades population, and the extent to which Oregon populations differ in genetic diversity, effective population size, and sensitivity to fragmentation is not well resolved from the available evidence (Good and Wake 1992; Miller et al. 2006; Wagner et al. 2006; Emel and Storfer 2015; Emel et al. 2019).

coastal tailed frog (*Ascaphus truei*)

Local abundance and spatial heterogeneity: Available information indicates that *A. truei* populations are spatially heterogeneous across their range, with substantial variation in occupancy and local abundance among watersheds and stream networks. Rangewide population size and long-term trends remain unknown, and inference is complicated by imperfect detectability and strong life-stage bias in survey observations. Many surveys disproportionately detect larvae and metamorphs, which experience high mortality and therefore provide limited insight into adult population size or long-term population stability (Hayes and Quinn 2015; Chelgren and Adams 2017).

Local abundance varies widely among occupied sites. In Oregon, some populations persist at consistently low densities over decades (e.g., Bull Run Lake; Corkran 2012), whereas others exhibit very high larval abundances where local habitat conditions are favorable. For example, Bury and Adams (1999) recorded hundreds of larvae and metamorphs across multiple Coast Range streams, and Chelgren and Adams (2017) captured and marked approximately 3,500 tadpoles along the Trask River, illustrating the potential for locally dense larval populations. Across studies, populations are typically numerically dominated by larvae, with substantially fewer metamorphs, juveniles, and adults detected (Adams 1993; Bury and Adams 1999; Hayes et al. 2006; Kroll et al. 2008; Corkran 2012).

Demographic and life history variation: Demographic information for *A. truei* indicates a life history characterized by prolonged larval development and delayed recruitment to the adult population. Larval periods typically range from approximately one to four years, depending on local thermal and hydrologic conditions (Hayes and Quinn 2015). Reproductive output is relatively low for an anuran, with clutch sizes typically on the order of several dozen eggs; estimates based on dissections of gravid females and hormone-induced ovulation suggest mean egg complements of approximately 57 eggs (range roughly 28–98), while field-observed clutches are often smaller (Karraker et al. 2006). Metamorphs and juveniles experience high mortality, and adults are infrequently detected in most survey efforts, contributing to uncertainty in estimates of adult abundance, survival, and population turnover (Hayes et al. 2006; Chelgren and Adams 2017). Life-stage composition within populations is therefore strongly skewed toward larvae, with adults representing a small and cryptic component of the population. This life-history structure limits inference about effective population size and demographic stability based solely on larval abundance, particularly in the absence of long-term mark–recapture data for adult frogs (Hayes and Quinn 2015).

Population genetics: Phylogeographic analyses of mitochondrial DNA reveal a deep divergence between *A. truei* and its congener *A. montanus*, which occurs further inland to the Rocky Mountains. This divergence is interpreted as the result of long-term isolation following uplift of the Cascade Mountains during the late Miocene to early Pliocene and supports recognition of the two taxa as separate species rather than population structure within *A. truei* itself (Nielson et al. 2001).

Oregon-specific genetic structure within *A. truei* has not been formally evaluated. The species' long larval period (Hayes and Quinn 2015), strong headwater habitat specialization, and limited routine adult movement suggest the potential for restricted dispersal and pronounced population structure across its range (Daugherty and Sheldon 1982; Hayes et al. 2006). However, genetic evidence indicates that populations are not strictly confined to stream networks. In Olympic National Park and Olympic National Forest, Spear and Storfer (2008) documented high genetic connectivity across unconnected river basins, providing evidence for population exchange via terrestrial movements. Closed forest cover and low solar radiation were positively associated with gene flow, suggesting that intact forested landscapes facilitate overland dispersal (Spear and Storfer 2008). Field observations further indicate that metamorphs, juveniles, and adults can make short terrestrial movements (generally <100 m) into or across stream-adjacent habitats during cool, wet conditions (Hayes and Quinn 2015).

Fine-scale genetic structure within stream networks can nevertheless be strongly altered by forest management. In a paired comparison of adjacent streams in coastal British Columbia, Wahbe et al. (2005) found significantly lower genetic diversity (expected heterozygosity reduced by >25%) in *A. truei* larvae from a clearcut stream relative to an old-growth stream within the same watershed. Larvae in the clearcut exhibited no decline in genetic similarity with increasing distance along a 180-m stream transect, whereas larvae in the old-growth stream showed a strong pattern of isolation by distance, consistent with localized reproduction and limited downstream mixing. These patterns were interpreted as evidence of a recent genetic bottleneck (i.e., loss of genetic diversity due to population contraction) or founder event (i.e., establishment by a limited number of individuals) following logging, coupled with altered dispersal behavior and reduced effective population size in the clearcut population (Wahbe et al. 2005).

In contrast, population genetic analyses conducted near the southern extent of the species' range in Mendocino County, CA, found no consistent evidence of recent genetic bottlenecks despite extensive, long-term forest management (Aguilar et al. 2013). Rather, populations exhibited strong genetic structuring primarily at the watershed scale, retained moderate levels of genetic diversity, and appeared to be at mutation–drift equilibrium. In this region, genetic connectivity was more strongly associated with the presence of cool, moist refugia and stream-connected habitats than with forest age or harvest history (Aguilar et al. 2013). These results indicate that genetic responses of *A. truei* populations to forest management are not uniform across the species' range and may differ substantially depending on regional climate, disturbance history, and landscape context.

Population diversity data gaps: Rangewide population size and long-term trends for *A. truei* remain unknown. Inference is constrained by imperfect detectability and strong life-stage bias in most surveys, which disproportionately detect larvae and metamorphs rather than adults. Because larval abundance can be locally high while adult detections remain low, larval counts alone provide limited insight into effective population size, adult survival, and long-term demographic stability, particularly in the absence of long-term adult mark–recapture data (Hayes and Quinn 2015; Hayes et al. 2006; Chelgren and Adams 2017).

Although local abundance varies widely among Oregon watersheds, including examples of persistent low-density populations and sites with very high larval densities, standardized monitoring sufficient to evaluate temporal trends, population persistence, and comparability among watersheds remains limited in the available evidence (Corkran 2012; Bury and Adams 1999; Chelgren and Adams 2017). Demographic parameters for later life stages remain largely unquantified.

Phylogeographic analyses support species-level divergence between *A. truei* and *A. montanus*, but fine-scale population genetic structure within Oregon has not been evaluated. As a result, genetic differentiation among Oregon watersheds, elevations, or ecoregions remains uncertain. While genetic evidence from the Olympic Peninsula indicates that populations may exchange individuals across watersheds in intact forested landscapes, the extent to which similar patterns occur in Oregon is unknown without targeted, Oregon-focused genetic sampling (Nielson et al. 2001; Spear and Storfer 2008).

Habitat Associations

Stream-associated amphibians are habitat specialists whose occupancy and distribution are constrained by local hydrologic, thermal, and structural conditions, particularly within headwater stream networks. This section synthesizes documented associations between the five focal amphibian species and key stream, microhabitat, and riparian forest features, drawing on peer-reviewed studies and agency syntheses. Emphasis is placed on habitat variables that have been empirically linked to occurrence or relative abundance, including stream size and permanence, channel gradient, substrate composition, hydrologic refugia, riparian canopy cover, and water temperature (e.g., Olson et al. 2007; Martin et al. 2021). Across stream-associated amphibians, channel gradient is frequently identified as an important habitat correlate, reflecting its influence on substrate stability, reduced retention and increased flushing of fine sediments, and hydrologic energy in cold, headwater channels, although the strength and consistency of this relationship vary among taxa and study contexts (Kroll 2009). In addition to physical habitat features, biotic context, including the presence of fish or crayfish, has been examined as a potential correlate of occupancy for some stream-associated amphibians. Crayfish presence shows a strong negative association with tailed frog (*Ascaphus truei*) occupancy, whereas fish presence has been discussed as a secondary, context-dependent contributor to observed occupancy patterns for torrent salamanders (*Rhyacotriton* spp.), acting largely in conjunction with stream size and channel gradient; however, in both cases, mechanistic roles remain incompletely resolved (Kroll et al. 2008; Kroll 2009).

Across taxa, all five species are most frequently associated with cold, shaded headwater environments characterized by coarse substrates and persistent surface or subsurface moisture, although the degree of specialization and tolerance varies among species. Torrent salamanders (*R. kezeri* and *R. variegatus*) represent the most microhabitat-restricted taxa, typically occupying seeps, splash zones, and low-order headwater channels where stable moisture conditions are maintained. While surface-flow intermittency is generally associated with reduced abundance, both species may persist in short, wetted reaches within otherwise intermittent channels, particularly where channel-connected seeps or subsurface flow provide localized hydrologic refugia (Ojala-Barbour et al. 2024). *Dicamptodon copei* shows similarly strong associations with cool, steep-gradient headwaters and coarse substrates, whereas *D. tenebrosus* exhibits greater ecological flexibility, occurring across a broader range of stream orders and aquatic habitat types, including larger streams and groundwater-influenced reaches. In contrast, *A. truei* is tightly associated with fast-flowing, high-gradient cobble and bedrock channels with permanent surface flow, reflecting its obligate reliance on lotic conditions throughout larval development.

Although numerous studies have examined relationships between amphibian occurrence or relative abundance and stream or riparian conditions, generalization across regions is limited by differences in study design, spatial scale, and analytical approaches. Across taxa, most habitat associations summarized below are derived from site- or reach-scale studies conducted within a limited number of watersheds. As a result, inference about habitat suitability is strongly scale-dependent, and relationships observed at local scales may not generalize across entire headwater networks or management-relevant spatial extents. Differences in sampling design, spatial coverage, and analytical treatment of imperfect detection further constrain direct comparison among studies and taxa. Accordingly, habitat associations are presented as relative patterns supported across available studies, rather than as definitive thresholds or prescriptions expected to apply uniformly across Oregon's managed forest landscapes.

Quantitative thresholds for many key habitat variables (e.g., channel gradient, canopy cover, flow permanence, temperature) remain poorly constrained across the entire range of these species. *R. kezeri* represents a partial exception, as recent habitat-suitability modeling quantified relationships between occupancy and hydrologic, topographic, and canopy variables across both reach and watershed scales (Thurman et al. 2025), providing empirical support for patterns previously described qualitatively. For the remaining taxa, habitat–landscape relationships are documented more inconsistently and are often derived from single-region studies, underscoring the need for caution when extrapolating across Oregon's managed forest landscapes. Accordingly, habitat associations summarized below emphasize relative patterns observed across studies, rather than definitive thresholds applicable across watersheds or management contexts.

Below, habitat associations are outlined for each species across three consistent dimensions: stream characteristics, microhabitat associations, and riparian forest context. These summaries draw on available evidence for each taxon. For each species, we also identify key data gaps and sources of uncertainty that limit inference at watershed or management-relevant scales. To facilitate comparison across species and to emphasize areas of convergence and divergence in habitat use, Table 4 summarizes these local habitat associations across the same three domains. This framework is intended to synthesize current evidence rather than define rigid habitat thresholds.

Table 4. Habitat association evidence framework.

Species	Stream characteristics	Microhabitat associations	Riparian forest context
Cope's giant salamander (*Dicamptodon copei*)	Cool, perennial 1st–3rd order headwater streams; most frequent in small, high-gradient channels; reduced abundance in intermittently dry reaches; neoteny common in cold or high-elevation streams	Pools and low-velocity habitats; coarse substrates (cobble, gravel interstices), under rocks, banks, and woody debris; cold summer temperatures	Shaded streams in coniferous or mixed forests; riparian canopy influences stream temperature and moisture; weak direct association with forest type or stand age
coastal giant salamander (*Dicamptodon tenebrosus*)	Broad distribution across headwater networks; common in 2nd–3rd order streams; occurrence varies with substrate and geology; abundance declines with channel drying and increases in groundwater-influenced reaches	Coarse substrates (especially boulders); pools and shaded reaches; abundance related to pool area and substrate size; sensitive to warming	Persists across somewhat broader range of forest ages than other NW stream associated amphibians, including managed forests; associations mediated through riparian shading, moisture, and channel stability rather than stand age
Columbia torrent salamander (*Rhyacotriton kezeri*)	Cold, low-order headwater streams with persistent surface flow or subsurface moisture; highest occurrence in steep-gradient channels with stable flow; absent from larger streams; abundance often highest near channel origins.	Spring-fed channels, seeps, and splash zones with coarse, permeable gravel–cobble substrates; low fine-sediment accumulation; strong association with hydrologic refugia and continuous surface–subsurface flow.	Shaded, moist headwater environments in coniferous forests; occupancy mediated primarily by riparian shading and hydrologic buffering; abiotic landform features (e.g., gradient, geology) often as important as forest age.
southern torrent salamander (*Rhyacotriton variegatus*)	Cold, low-order headwater streams with persistent surface or subsurface moisture; highest abundance in steep, high-relief channels with coarse substrates; reduced occurrence with warming or seasonal drying.	Spring-fed channels, seeps, and splash zones with coarse gravel–cobble substrates and abundant interstitial spaces; larvae aquatic in percolating gravels, adults concentrated in splash zones and turbulent stream margins.	Shaded, moist headwater environments with intact riparian canopy; higher densities in uncut forests; forest age possibly effects variable and mediated through shading, moisture, and sediment control.
coastal tailed frog (*Ascaphus truei*)	Cold, fast-flowing, shaded perennial headwater streams; breeding and development restricted to permanently flowing channels with coarse substrates; patchy occupancy influenced by flow permanence and sediment inputs	Larvae cling to cobble and bedrock in riffles and cascades; adults use riffles, pools, and waterfall margins; sensitive to fine sediment and elevated temperatures during early life stages	Occurs most frequently in older forests but persists in managed stands where riparian shading, coarse substrates, and permanent flow remain intact; forest age effects mediated by hydrologic and thermal context

Cope's giant Salamander (*Dicamptodon copei*)

Stream characteristics: *D. copei* is restricted to cool, perennial first- through third-order headwater streams, most commonly occurring in smaller, high-gradient channels within forested landscapes (Adams and Bury 2002; Roni 2002; Kroll et al. 2010; Foster and Olson 2014). The species is frequently associated with steep reaches and is rarely documented in intermittent or low-gradient streams. Within occupied streams, *D. copei* is often found in pool habitats and other low-velocity areas rather than in fast-flowing riffles (Foster et al. 2015). In a study of both *D. copei* and *D. tenebrosus*, Ojala-Barbour et al. (2024) found that relative abundance was lower at the very upper ends of headwater streams and in first-order reaches, increased slightly downstream within headwater channels, declined sharply where channels dried, and was higher in reaches influenced by groundwater seeps. Neoteny is common, particularly in colder or higher-elevation streams, allowing populations to persist permanently in aquatic environments (Nussbaum et al. 1983; Wagner 2015).

Microhabitat associations: Individuals occupy coarse-substrate microhabitats within streams, including interstitial spaces among cobble and gravel, areas beneath rocks, undercut banks, and in-stream coarse woody debris, which provide cover and foraging opportunities (Nussbaum et al. 1983; Adams and Bury 2002) In Olympic Peninsula streams, *D. copei* abundance was highest at intermediate levels of in-channel coarse woody debris, with densities peaking near ~10% substrate coverage (Adams and Bury 2002). Stream temperatures at occupied sites are characterized by cold water temperatures, with field observations typically reporting summer temperatures ranging from approximately 7.5–13.0 °C and mean values near 11 °C (Foster 2015), but laboratory studies report physiological stress responses at temperatures ≥21 °C (70 °F) (Wagner 2015).

Riparian forest context: *D. copei* is associated with shaded streams in coniferous or mixed forests, primarily through the influence of riparian canopy on stream temperature and moisture regimes rather than through direct associations with specific forest types or stand ages (Adams and Bury 2002; Steele et al. 2002; Bisson et al. 2002). Reductions in canopy cover can increase stream temperatures beyond tolerated levels and may degrade habitat quality for this species (Bull and Wales 2001; Adams and Bury 2002).

Habitat data gaps: Evidence linking *D. copei* occurrence or relative abundance to stream gradient, groundwater influence, longitudinal position, and thermal regime is drawn from multiple independent studies; however, these relationships have rarely been evaluated using common analytical frameworks or consistently detection-corrected approaches. Quantitative limits for key variables, such as field-based thermal tolerances, frequency or duration of seasonal drying, and minimum stream size supporting population persistence, remain poorly constrained, limiting cross-watershed comparison and management inference.

Evidence for microhabitat associations is similarly based on descriptive or site-specific analyses, with limited evaluation of how variation in substrate composition, coarse woody debris, or habitat complexity influences occupancy or abundance beyond individual study sites. As a result, the generality of reported microhabitat relationships across broader spatial extents remains uncertain.

Riparian influences on *D. copei* habitat use are inferred largely through effects on stream temperature and moisture rather than through direct relationships with forest composition or stand age. Although canopy reduction is known to elevate stream temperatures beyond tolerated levels, few studies explicitly integrate riparian forest attributes with in-stream habitat conditions or population responses at watershed scales. Consequently, the role of riparian structure in mediating habitat suitability for *D. copei* across Oregon landscapes remains incompletely characterized.

Although some studies have explicitly accounted for imperfect detection using occupancy or network-based modeling approaches (e.g., Kroll et al. 2008; Ojala-Barbour et al. 2024), many investigations relating *D. copei* occurrence or abundance to stream and microhabitat features rely on uncorrected presence or relative abundance data, constraining inference about the strength and comparability of reported habitat relationships.

coastal giant salamander (*Dicamptodon tenebrosus*)

Stream characteristics: *D. tenebrosus* occurs across a broader portion of stream networks, occurring across a wider range of conditions than many other headwater stream-associated amphibians (Olson and Weaver 2007), and is often the dominant amphibian in second- and third-order streams across mixed conifer–hardwood forests (Welsh and Lind 2002). Kroll et al. (2008) sampled 141 perennial streams across commercial forest landscapes in western Oregon and Washington and found that detection-corrected occupancy of *Dicamptodon* spp. (*D. tenebrosus* and *D. copei*) varied with stream substrate type, with higher occupancy in streams associated with consolidated geologies; however, analyses did not resolve quantitative thresholds for stream size, gradient, or temperature. Although *D. tenebrosus* is not confined to steep-gradient headwaters and has been documented across a range of stream orders and channel conditions, stream gradient remains an important predictor of local presence and abundance, likely reflecting its role in maintaining coarse substrates and facilitating the flushing of fine-sediments (Kroll 2009; Dudaniec and Richardson 2012). Furthermore, its association with stream gradient likely reflects the combined influence of physical habitat structure and biotic context, including prey availability and interactions with competitors or predators, rather than gradient alone (Neal et al. 2024). In a study of both *D. tenebrosus* and *D. copei*, Ojala-Barbour et al. (2024) found that relative abundance increases with higher stream order, is reduced in the uppermost headwater reaches, and declines sharply with channel drying, but is elevated in reaches influenced by groundwater seeps.

Microhabitat associations: Relative abundance has been consistently associated with coarse substrate composition, particularly the proportion of boulders, which highlights the importance of physical stream structure at the reach scale (Dudaniec and Richardson 2012). The species is associated with pools, coarse substrates, and narrow, shaded stream reaches (Welsh and Lind 2002). Reach-scale studies in fish-bearing headwater streams of the northern Oregon Coast Range documented consistent occupancy in second-growth forests, with abundance and biomass best explained by pool area and substrate size, not canopy cover, wood volume, or fish density (Neal et al. 2024). Experimental flow-reduction and warming manipulations further support the importance of local hydrologic conditions for *D. tenebrosus*. In a paired before–after control–impact experiment in a second-order stream, Maffia et al. (2025) found that diverting ~50% of streamflow reduced wetted habitat and pool area, yet salamander abundance declined less in the low-flow reach than in a warmed reach subjected to modest temperature increases (~0.41–0.63 °C; ~0.7–1.1 °F), when evaluated relative to an unmanipulated reference reach. Based on larval individuals sampled from forested streams in the western Oregon Cascades, coastal giant salamanders exhibit critical thermal maxima near 29 °C (84 °F) (Bury 2008).

Riparian forest context: *D. tenebrosus* can persist in managed and second-growth forest landscapes, reflecting ecological plasticity (Olson and Weaver 2007), and occurs across a wide range of forest ages, with forest age alone being a poor predictor of density or occurrence (Welsh and Lind 2002). At the stand scale, occupancy of *Dicamptodon* spp. was found to increase with stand age in commercial forests, although stand age was treated as a coarse correlate rather than a mechanistic driver, and detection probabilities <1 introduce uncertainty into the inferred relationship (Kroll et al. 2008). Even following high-severity wildfire and post-fire salvage, densities of this species are more closely linked to landscape position than to riparian disturbance in the short term (Swartz et al. 2025).

The species is closely associated with moist riparian environments, and shaded conditions likely play an important role in maintaining suitable thermal and hydrologic conditions, even where populations occur in harvested or regenerating forests. In British Columbia, at the northern range margin of this species, relative abundance increases with forest age, likely through indirect effects on stream stability, sediment dynamics, and substrate availability rather than direct effects of overstory conditions alone (Dudaniec and Richardson 2012). Retention of intact riparian forest conditions is likely important for maintaining suitable microclimates and for sustaining terrestrial movement pathways used by adult *D. tenebrosus*, with genetic evidence suggesting movement contributes to connectivity among headwater drainages (Olson et al. 2007; Johnston and Frid 2002; Auteri et al. 2022).

Habitat data gaps: Evidence linking *D. tenebrosus* occurrence or relative abundance to stream characteristics, including stream order, channel gradient, substrate composition, groundwater influence, and local hydrologic conditions, is supported across multiple independent studies. However, quantitative thresholds for key variables such as temperature limits, drying frequency, substrate composition, and pool availability remain poorly constrained, limiting comparison across watersheds and management contexts.

Evidence for microhabitat associations is similarly derived largely from reach-scale or site-specific studies. While relationships with coarse substrates, pool area, and local hydrologic conditions are consistently reported, few studies have evaluated how variation in these features influences occupancy or abundance across broader spatial extents. As a result, the generality of reported microhabitat relationships beyond individual study systems remains uncertain.

Riparian influences on *D. tenebrosus* habitat use are inferred primarily through effects on stream shading, moisture, channel stability, and sediment dynamics rather than through direct relationships with forest age or composition (Welsh and Lind 2002; Olson and Weaver 2007). Reported relationships vary regionally, including patterns observed near the northern range margin of the species (Dudaniec and Richardson 2012), and few studies explicitly integrate riparian forest attributes with in-stream habitat conditions or population responses at watershed scales. Consequently, uncertainty remains in how forest structure mediates habitat suitability for *D. tenebrosus* across Oregon landscapes.

Although some studies have explicitly accounted for imperfect detection using occupancy or network-based modeling approaches (e.g., Kroll et al. 2008; Ojala-Barbour et al. 2024), many investigations relating *D. tenebrosus* occurrence or abundance to stream and microhabitat features rely on uncorrected presence or relative abundance data. This heterogeneity in analytical approaches constrains inference about the strength, consistency, and comparability of reported habitat relationships.

Columbia torrent salamander (*Rhyacotriton kezeri*)

Stream characteristics: *R. kezeri* is a headwater obligate restricted to cold, perennial streams, seeps, and splash zones in small montane watersheds (Good and Wake 1992; Steele et al. 2003; Russell et al. 2004). Occupied streams with higher densities are typically low-order channels with steep gradients (e.g., >20%) and stable perennial flows (Wilkins and Peterson 2000; Russell et al. 2004; Kroll et al. 2008). The species is absent from larger streams and rivers and remains closely tied to headwater environments at the watershed scale (Good and Wake 1992), with no detections in watersheds >13.5 ha in a SW Washington study (Wilkins and Peterson 2000). While fish presence was not a top-ranked covariate in occupancy models for torrent salamanders in 1st–3rd order stream segments, Kroll et al. (2008) suggested that trout in low-gradient reaches may contribute to observed gradient–occupancy relationships, emphasizing that such patterns should be interpreted cautiously given the descriptive nature of occupancy models. Although surface flow intermittency reduces relative abundance, *R. kezeri* may seasonally persist in short, wetted reaches within predominantly dry headwater channels where channel-connected seeps or subsurface flow provide localized moisture refugia (Ojala-Barbour et al. 2024). Across Oregon and Washington forests, occupancy is consistently and positively associated with channel gradient and, in some landscapes, basalt-dominated geology, which supports coarse substrates and cold, well-oxygenated flows (Russell et al. 2004; Kroll et al. 2008). In the Oregon Coast Range, Columbia torrent salamanders have been documented consistently in streams with summer temperatures ≤12 °C (54 °F) (Russell et al. 2004).

Rangewide habitat-suitability modeling identifies baseflow stability, summer moisture balance, and canopy cover as important predictors of occurrence (Thurman et al. 2025), indicating that broader hydrologic and climatic settings influence where this species is found. Within headwater systems, abundance is often greatest near channel origins and uppermost reaches, declining downstream even where streams remain perennial (Russell et al. 2004; Wilkins and Peterson 2000). Within occupied streams, *R. kezeri* exhibits pronounced patchy distribution, with individuals absent from substantial portions of otherwise suitable channels, particularly in lower-gradient reaches (Russell et al. 2004). Habitat associations for *R. kezeri* have been shown to be strongly scale-dependent, where landscape-level patterns reflect broad landform and hydrologic constraints but stream- and reach-scale occupancy is more variable and may be underestimated when surveys are confined to short, randomly selected reaches (Russell et al. 2004).

Microhabitat associations: At the reach scale, *R. kezeri* is most frequently occurs in spring-fed channels, seeps, and splash zones where substrates are coarse and permeable and water remains in continuous contact with saturated surfaces (Good and Wake 1992; Welsh and Lind 1996; Russell et al. 2004). Across multiple studies, higher salamander abundance has been associated with streambeds dominated by gravel or cobble with minimal fine sediment accumulation, particularly in higher-gradient reaches where fine sediments are less likely to accumulate (Wilkins and Peterson 2000; Russell et al. 2004; Stoddard and Hayes 2005; Kroll et al. 2008). In the Oregon Coast Range, both Columbia and southern torrent salamanders showed reduced occurrence in reaches with a high proportion of small (<32 mm) substrate (Stoddard and Hayes 2005). By contrast, in southwestern Washington, the presence of Columbia torrent salamanders was unrelated to substrate size or underlying lithology, and instead was more closely linked to local topographic conditions, including steep channel gradients, small basin areas, and north-facing aspects associated with cooler, wetter microclimates (Wilkins and Peterson 2000). Relative abundance is strongly associated with hydrologic refugia, particularly channel-connected seeps, and declines with increasing channel drying (Ojala-Barbour et al. 2024).

Riparian forest context: *R. kezeri* are frequently associated with mature or late-successional coniferous forests that maintain cool, humid microclimates and stable headwater hydrology (Nussbaum et al. 1983; Bury and Corn 1988; Corn and Bury 1989). However, forest age alone is not a sufficient predictor of occupancy. In a study across Oregon and Washington, a quadratic relationship with stand age was found across *Rhyacotriton* spp, with lowest occupancy in recent clearcuts and highest occupancy in intermediate-aged forests, particularly where steep gradients and favorable geologic conditions were present (Kroll et al. 2008). Several studies indicate that abiotic landform features explain occupancy as well as or better than vegetation age, suggesting that topographic context and geomorphic controls on hydrology and microclimate (e.g., stream size and gradient, headwater position, and aspect) are more influential than stand age per se (Russell et al. 2004; Olson and Ares 2022).

Habitat data gaps: Evidence linking *R. kezeri* occurrence or abundance to stream characteristics, including channel gradient, stream size, substrate composition, summer temperature, and hydrologic stability, is supported across multiple studies or sites. However, quantitative thresholds for key variables, such as tolerance to seasonal drying, minimum stream size, and field-based thermal limits, could be further constrained, limiting comparison across watersheds and management contexts.

Evidence for microhabitat associations is similarly derived largely from reach-scale or site-specific studies. While strong relationships with coarse, permeable substrates and hydrologic refugia (e.g., channel-connected seeps and persistent subsurface flow) are consistently reported, fewer studies have evaluated how variation in these features influences occupancy or abundance across broader spatial extents. As a result, the generality of reported microhabitat relationships beyond individual study systems remains uncertain.

Associations with riparian forest context are supported primarily through inferred effects on topographic configuration, microclimate regulation, and hydrologic buffering rather than through direct effects of forest age alone (Russell et al. 2004; Olson and Ares 2022). Although non-linear relationships between stand age and occupancy have been documented in some landscapes (Kroll et al. 2008), few studies explicitly integrate riparian forest attributes with in-stream or microhabitat conditions at watershed scales. Consequently, uncertainty remains in how forest structure mediates habitat suitability for *R. kezeri* across Oregon landscapes.

Although some studies have explicitly accounted for imperfect detection using occupancy-based approaches (e.g., Kroll et al. 2008), many investigations relating occurrence or abundance to stream and microhabitat features rely on uncorrected presence or relative abundance data. This heterogeneity in analytical treatment constrains inference about the strength, consistency, and comparability of reported habitat relationships across spatial contexts.

southern torrent salamander (*Rhyacotriton variegatus*)

Stream characteristics: *R. variegatus* occupies cold, small headwater streams, seeps, and springs, including channels with seasonally intermittent surface flow but persistent subsurface moisture, and is absent from larger stream channels (Good and Wake 1992; Welsh and Lind 1996; Martin and McComb 2003; Tait and Diller 2006; Olson and Ares 2022). These small streams serve as breeding, larval, and dispersal habitat within watersheds (Good and Wake 1992; Emel and Storfer 2015). Abundance and occupancy in both adult and juvenile torrent salamanders are positively associated with higher stream gradients (Diller and Wallace 1996; Wilkins and Peterson 2000; Stoddard and Hayes 2005; Kroll et al. 2008). Although fish presence was not a top-ranked covariate in occupancy models for torrent salamanders in 1st–3rd order stream segments, Kroll et al. (2008) suggested that trout in low-gradient reaches may contribute to observed gradient–occupancy relationships, emphasizing that such patterns should be interpreted cautiously given the descriptive nature of occupancy models. Southern torrent salamanders exhibit low thermal tolerance relative to most amphibians, typically occurring in streams where summer temperatures are ≤15–16 °C (59–61 °F). Thermal tolerance estimates are based on laboratory and field data from populations sampled in western Oregon (Bury 2008). Highest abundances occur at stream temperatures generally <15 °C (59 °F), with signs of thermal stress near ~17 °C (80 °F) and reduced occurrence or abundance where warming or seasonal drying are present (Welsh and Lind 1996; Bury 2008).

Microhabitat associations: Larvae and adults occupy well-shaded seeps and stream margins characterized by cold, clear, well-aerated water (Good and Wake 1992; Welsh and Lind 1996). Larvae are entirely aquatic and may occur at high densities in gravel substrates with percolating water, whereas adults preferentially use large rocks and are most abundant in splash zones of waterfalls and turbulent stream margins (Good and Wake 1992; Welsh and Lind 1996). In the Oregon Coast Range, both southern and Columbia torrent salamanders showed reduced occurrence in reaches with a high proportion of small (<32 mm) substrate (Stoddard and Hayes 2005). Adults may occasionally move through moist upland riparian forest during wet conditions, but post-metamorphic individuals typically remain within a few meters of stream channels, with only rare movements documented far from surface water during wet conditions (Welsh and Lind 1996; Leppin et al. 2020). *R. variegatus* avoids open water and strongly prefers gravel, pebble, and cobble substrates that provide cover and interstitial spaces for foraging and predator avoidance (Good and Wake 1992; Welsh and Lind 1996; Stoddard and Hayes 2005). Optimal microhabitats contain a diversity of coarse particle sizes, abundant interstitial crevices, and variable flow conditions (Welsh and Lind 1996).

Riparian forest context: *R. variegatus* is strongly associated with intact riparian canopy cover, which moderates stream temperature, maintains humidity, and limits sediment inputs. Ideal habitat conditions include forests with large conifers, abundant moss, and high canopy closure characteristic of late-seral stands (Welsh and Lind 1996; Nussbaum et al. 1983). In western Oregon, densities and biomass were significantly higher in uncut forests compared with logged forests (Corn and Bury 1989), While accounting for imperfect detection, Kroll et al. (2008) found that occupancy across *Rhyacotriton* species on managed forests in western Oregon and Washington exhibits a quadratic relationship with stand age, peaking in stands of intermediate-age and declining in both younger and older age classes. Regardless of forest age, downed wood and forest cover provide the cool, moist microclimate required for terrestrial activity and short-distance dispersal (Emel et al. 2019; Olson and Burton 2019). In the northern portion of the species' range, forest fragmentation has been associated with increased genetic differentiation among populations (Emel et al. 2019).

Habitat data gaps: Evidence linking *R. variegatus* occurrence or abundance to stream characteristics, including channel gradient, stream size, substrate composition, summer temperature, and flow permanence, is supported across multiple studies. However, quantitative thresholds for key variables, such as tolerance to seasonal drying, thermal exposure under field conditions, and fine-sediment accumulation, remain unconstrained, limiting comparison across watersheds and management contexts.

Evidence for microhabitat associations is similarly derived largely from reach-scale or site-specific studies. While strong relationships with coarse gravel–cobble substrates, abundant interstitial spaces, and splash-zone or seep habitats are consistently reported, fewer studies have evaluated how variation in microhabitat structure and sediment conditions influences occupancy or abundance across broader spatial extents. As a result, the generality of reported microhabitat relationships beyond individual study systems remains uncertain.

Associations with riparian forest context are supported primarily through inferred effects on stream shading, microclimate regulation, and sediment control rather than through direct effects of forest age alone (Welsh and Lind 1996; Olson and Ares 2022). Although higher densities and biomass have been documented in uncut forests (Corn and Bury 1989), and non-linear relationships between stand age and occupancy have been reported across *Rhyacotriton* species (Kroll et al. 2008), few studies explicitly integrate riparian forest attributes with in-stream or microhabitat conditions at or across watershed scales. Consequently, uncertainty remains in how forest structure mediates habitat suitability for *R. variegatus* across Oregon landscapes.

Although some studies have accounted for imperfect detection using occupancy-based approaches (e.g., Kroll et al. 2008), many investigations relating occurrence or abundance to stream and microhabitat features rely on uncorrected presence or relative abundance data. This heterogeneity in analytical treatment constrains inference about the strength, consistency, and comparability of reported habitat relationships across spatial contexts.

coastal tailed frog (*Ascaphus truei*)

Stream characteristics: *A. truei* is strongly associated with cold, fast-flowing, shaded perennial headwater streams and are commonly described as headwater obligates, breeding and developing almost exclusively within small, permanently flowing channels (Hayes et al. 2006; Hayes and Quinn 2015). They are highly aquatic, sensitive to dehydration, and require a narrow thermal range (approximately 5–18.5 °C; 41-64 °F), remaining in or near water throughout their life cycle (Hayes and Quinn 2015; Washington Herp Atlas 2017). Occupied streams typically occur in catchments exceeding ~50 ha and are at least ~1 m wide (Hayes et al. 2006; Kroll et al. 2010). Occupancy is patchy within and among watersheds, reflecting the species' narrow hydrologic and substrate requirements and sensitivity to local habitat conditions, including flow permanence and sediment inputs (Hayes and Quinn 2015; Welsh and Lind 2002; Stoddard and Hayes 2005). Long larval periods (1–4 years) further constrain the species to cool, well-oxygenated streams with permanent hydroperiods (Hayes and Quinn 2015). Detection-corrected occupancy modeling across commercial forests in Oregon and Washington further indicates that *A. truei* occupancy is negatively associated with the presence of crayfish and exhibits a quadratic relationship with bankfull channel width (lowest at small and large widths) (Kroll et al. 2008).

Microhabitat associations: Adults use a variety of stream mesohabitats, including riffles, pools, and waterfall margins, while larvae cling to cobble and bedrock surfaces in high-gradient channels for foraging and refuge and are sensitive to increased fine sediment loads that reduce interstitial space (Stoddard and Hayes 2005; (Adams 1993; Stoddard and Hayes 2005; Karraker et al. 2006; Hayes and Quinn 2015). These coarse substrates provide refugia and turbulent flow conditions that reduce predation risk from stream-dwelling salamanders (Welsh and Lind 2002). In experiments that used larvae from the Oregon Coast Range, results showed that larvae select coarse instream substrates dominated by gravel and cobble (approximately 55–125 mm diameter) (Altig and Brodie 1972; Hayes and Quinn 2015). Within occupied streams, larvae are more frequently associated with shallow, turbulent microhabitats such as riffles, cascades, than with pools (Welsh and Lind 2002). Reproductive activity and early development are closely tied to cold-water conditions: oviposition typically occurs near ~11 °C (52 °F), larval development is optimized near ~12 °C (52 °F), and upper thermal limits are near ~18 °C (64 °F) (Adams 1993; Karraker et al. 2006). Estimates derived from populations sampled across the Pacific Northwest show that coastal tailed frogs exhibit low thermal tolerance relative to most amphibians, with larvae and adults experiencing physiological stress at temperatures above approximately 18–22 °C (64–72 °F) (Bury 2008).

Riparian forest context: Detections of *A. truei* are often more frequent in older forest stands (generally >~100 years), particularly where mature riparian canopies maintain cool, shaded stream conditions and stable coarse substrates (Welsh and Lind 2022). However, the species also occurs in younger managed forests when key riparian and instream attributes remain intact, such as canopy cover, stream shading, and coarse substrate availability (Stoddard and Hayes 2005; Kroll et al. 2008; Hayes and Quinn 2015). Detection-corrected occupancy modeling across commercial forests in Oregon and Washington indicates a positive but non-exclusive association between *A. truei* occupancy and stand age, with substantial occupancy observed in younger stands, particularly in the absence of crayfish and under favorable stream conditions (Kroll et al. 2008). Collectively, these findings indicate that forest seral stage alone is not a consistent predictor of occupancy; instead, climatic context and hydrologic conditions appear to mediate observed associations with older forests, particularly at lower latitudes and interior locations (Hayes and Quinn 2015; McEwan 2014; Halofsky et al. 2022).Maintenance of cool stream temperatures, low sediment loads, and permanent flow appears more important than stand age per se for supporting occupancy and reproduction (Karraker et al. 2006; Hayes and Quinn 2015). In a long-term before–after, control–impact experiment, coastal tailed frog densities did not differ at 2-years post-harvest, but larval and post-metamorphic densities were lower 7–8 years after forest harvest across all riparian buffer treatments, including continuous buffers (McIntyre et al. 2025).

Habitat data gaps: Evidence linking *A. truei* occurrence to stream characteristics, including stream size, channel gradient, substrate composition, flow permanence, and thermal regime, is supported across multiple studies. However, quantitative thresholds for key variables, particularly tolerance to fine sediment, thermal exposure under field conditions, and hydrologic alteration, remain poorly constrained, limiting comparison across watersheds and management contexts.

Evidence for microhabitat associations is similarly derived largely from reach-scale or site-specific studies. While strong relationships with coarse substrates, turbulent flow conditions, and cold-water microhabitats are consistently reported for larvae and adults, relationships between microhabitat degradation and population persistence, especially across the species' prolonged larval period, have rarely been evaluated beyond local scales. As a result, uncertainty remains regarding how localized changes in substrate condition, sediment loading, or thermal regime translate into long-term population outcomes (Stoddard and Hayes 2005; Hayes and Quinn 2015).

Although occupancy is often higher in older forests, detection-corrected analyses indicate that forest seral stage alone is not a sufficient predictor of occupancy (Kroll et al. 2008; McIntyre et al. 2025), and few studies explicitly integrate riparian forest attributes with in-stream habitat conditions or population responses at watershed scales. Consequently, uncertainty remains in how riparian forest structure mediates habitat suitability and population persistence for *A. truei* across Oregon landscapes.

Although some studies have accounted for imperfect detection using occupancy-based approaches (e.g., Kroll et al. 2008), many investigations relating occurrence or abundance to stream and microhabitat features rely on uncorrected presence or relative abundance data. This heterogeneity in analytical treatment constrains inference about the strength, consistency, and comparability of reported habitat relationships across spatial contexts.

Torrent Salamander Population Status and Trends

Assessing population status and trends for torrent salamanders is important for evaluating the persistence of headwater-associated amphibians on private forestlands regulated under the Oregon Forest Practices Act. The Columbia torrent salamander (*Rhyacotriton kezeri*) and southern torrent salamander (*R. variegatus*) are cold-water specialists associated with perennial headwater streams, exhibit limited dispersal capacity, and occur in patchy distributions shaped by local hydrology, substrate, and riparian conditions. Following taxonomic revision by Good and Wake (1992), *R. kezeri* and *R. variegatus* are recognized as two of four genetically distinct torrent salamander species endemic to the Pacific Northwest, replacing the earlier treatment of all torrent salamanders as a single species (*R. olympicus*). *R. kezeri* is restricted to the northern Oregon Coast Range and adjacent Washington populations, whereas *R. variegatus* occupies a broader geographic range extending through western Oregon into northern California (see Distribution section).

Despite their conservation relevance, direct information on population size, demographic rates, or long-term trends is lacking for both species. Available evidence is derived primarily from studies reporting occupancy, detection frequency, and relative abundance at stream-reach or watershed scales across gradients of forest condition and hydrologic permanence. These studies provide insight into patterns of occurrence and localized persistence but do not support quantitative assessment of population trends at watershed or rangewide scales. Moreover, because changes in occupancy or detection do not necessarily reflect changes in population size, survival, or recruitment, observed patterns should not be interpreted as direct evidence of population increase or decline. Consistent with AMPC Question 2, this section evaluates available evidence bearing on population status and relative trends, while explicitly identifying limitations that preclude quantitative trend estimation. Because many of the limitations on inference reflect shared methodological constraints rather than species-specific differences, data gaps are summarized jointly for both torrent salamander species following the species summaries.

Columbia torrent salamander (*Rhyacotriton kezeri*)

Population status: Population size and long-term trends for *R. kezeri* remain unknown. At the stream scale, the species was detected in 58% of sampled first-order headwater streams in managed forests of Oregon's Coast Range (Russell et al. 2004) and in approximately 53% of sampled perennial headwater streams in managed forests of southwestern Washington (Wilkins and Peterson 2000). These estimates represent observed detections and do not explicitly account for imperfect detection. Within occupied streams, relative abundance estimates indicate substantial spatial variability. In Oregon, landscape-scale abundance ranged from 0 to 4.77 individuals per linear meter of stream (mean = 0.47 individuals/m), with larvae comprising 59.6% of captures (Russell et al. 2004). At finer spatial scales, randomly selected 10-m reach surveys detected salamanders in approximately half of sampled reaches and reported mean abundance of 2.24 ± 0.89 individuals/m^2, with larvae comprising 62.6% of captures. These reach-scale estimates similarly reflect indices of relative abundance rather than detection-corrected density.

Detection-corrected occupancy modeling in managed forests of Oregon and Washington reported high occupancy of torrent salamanders (*Rhyacotriton* spp.), exceeding 80% in stands of intermediate age and moderate channel gradient; however, estimates were not species-specific and combined *R. kezeri* with other *Rhyacotriton* taxa (Kroll et al. 2008). Recent rangewide habitat-suitability modeling relates observed occurrence of *R. kezeri* to climatic, hydrologic, and forest-structure variables, including stream gradient, summer water balance, streamflow permanence, and riparian canopy cover (Thurman et al. 2025). Although this modeling framework provides a consistent basis for comparing relative occurrence across watersheds, estimates are based on observed detections and do not explicitly model imperfect detection.

Trends and pressures: Quantitative population trend data are limited. Historical landscape modeling estimated substantial reduction in suitable habitat since Euro-American settlement, attributed primarily to riparian canopy loss and forest fragmentation (Kagan et al. 1999). Field studies and occupancy analyses consistently report higher occurrence where perennial headwater flow and riparian shading are retained, and lower detection probabilities in recently harvested or hydrologically altered headwaters (Bury and Corn 1988; Kroll et al. 2008; Olson and Ares 2022). However, these patterns describe variation in occurrence and habitat association rather than direct evidence of population decline.

southern torrent salamander (*Rhyacotriton variegatus*)

Population status: Population size and rangewide trends for *R. variegatus* remain unknown. At the stream scale, the species has been frequently detected in headwater streams with suitable habitat conditions. In managed forest landscapes of western Oregon, occupancy exceeding 80% was reported in 45-year-old stands with moderate channel gradients and basalt or mixed substrates; this estimate was derived from surveys that combined both *Rhyacotriton* species and did not distinguish between *R. variegatus* and *R. kezeri* (Kroll et al. 2008). In northern California, *R. variegatus* was detected in 80.3% of surveyed headwater streams on managed forest lands (Diller and Wallace 1996).

In contrast, surveys conducted at randomly selected sites within the Klamath Mountains documented substantially lower detection frequencies (28.2–36.8%), with detections increasing to 46.6–62.3% at sites meeting specific microhabitat criteria (Welsh and Lind 1992), indicating strong sensitivity of observed occupancy to site selection and habitat conditions. Reported densities vary widely with habitat quality and sampling design. In Lincoln County, Oregon, high larval densities (mean 12.9 larvae/m^2) were recorded in favorable headwater habitat (Nussbaum and Tait 1977). Across California, densities ranging from 1 to 50 individuals per 10 m^2 plot (mean 0.68 individuals/m^2) have been reported in suitable habitat (Welsh and Lind 1996), while lower mean densities (0.28 individuals/m^2) were documented in randomly selected managed forest streams (Diller and Wallace 1996). These estimates describe local relative abundance within surveyed reaches and do not provide estimates of population size beyond sampled areas. Watershed-scale surveys that explicitly accounted for imperfect detection reported near-ubiquitous occupancy of torrent salamanders (*Rhyacotriton* spp.) in third-order watersheds within intensively managed forests of Oregon and Washington, indicating that reach-scale absences may not reflect watershed-level absence and providing a baseline for future comparisons of relative occupancy (Kroll et al. 2009).

Trends and pressures: Quantitative population trend data are not available. Landscape-scale modeling indicates substantial historical loss of suitable habitat since Euro-American settlement, particularly in the Coast Range and Klamath Mountains (Kagan et al. 1999). Field studies consistently report reduced occupancy or abundance in recently harvested or hydrologically altered headwaters and higher persistence where perennial flow and riparian shading are retained (Corn and Bury 1989; Welsh and Lind 1996; Kroll et al. 2008; Olson and Ares 2022). Although some studies describe persistence or reappearance in managed landscapes as riparian conditions recover, direct evidence of recolonization dynamics remains limited.

Population data gaps (Columbia and southern torrent salamanders)

No time-series datasets currently exist to evaluate population trajectories, extinction risk, or recolonization dynamics for either *R. kezeri* or *R. variegatus* at watershed or rangewide scales. Available information for both species is derived primarily from single-period or short-term surveys conducted in limited portions of their respective ranges, restricting inference about temporal change. Most estimates of occupancy and abundance are based on observed detections and relative abundance indices rather than detection-corrected occupancy or density. An exception is watershed-scale surveys conducted in intensively managed forests of Oregon and Washington that explicitly accounted for imperfect detection and reported near-ubiquitous occupancy of torrent salamanders (*Rhyacotriton* spp.) in third-order watersheds, indicating that reach-scale absences may not reflect watershed-level absence and providing a baseline for future comparisons of relative occupancy (Kroll et al. 2008, Kroll et al. 2009). However, these estimates were not species-specific, were geographically limited, and do not include repeated measures necessary to assess trends through time. More recently, ongoing multi-year survey efforts in the northern Oregon Coast Range are beginning to resample standardized locations and incorporate detection-corrected occupancy approaches at watershed scales (Garcia et al. 2025); however, these efforts are still underway and thus not summarized here.

More broadly, although detection-corrected occupancy methods for stream-associated amphibians are well established and feasible at reach to watershed scales using repeat in-stream surveys, comparable approaches for estimating abundance, demographic rates, or population change at larger spatial extents remain limited. Capture-based abundance estimates are generally restricted to localized stream reaches, egg-based surveys are rarely feasible at scale, and emerging environmental DNA (eDNA) approaches show promise for improving detection but currently lack validated relationships with abundance or life-stage–specific demographic parameters. For *R. kezeri* specifically, recent rangewide habitat-suitability modeling provides a spatially consistent framework for assessing relative occurrence across its restricted geographic range (Thurman et al. 2025). While this work represents an important advance for spatial inference, it is based on observed detections and does not explicitly account for imperfect detection or temporal replication. Consequently, repeated surveys using comparable methods would be required to evaluate trends or changes in occupancy over time.

Overall, the absence of standardized, repeat-visit monitoring frameworks or hierarchical modeling approaches constrains the ability to distinguish true absence from non-detection, evaluate change over time, or compare estimates across forest management. In addition, while localized demographic and life-history studies exist and are summarized elsewhere in this review, these efforts are generally short-term and reach-scale and are not designed to support inference about population trajectories across watersheds or ownerships, particularly under variation in streamflow permanence and thermal conditions.

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Appendix A. AMPC Questions and Review Scope

Table A1. Adaptive Management Program Committee (AMPC) questions relevant to amphibians and the scope of this literature review. This table lists the AMPC questions developed under the Oregon Private Forest Accord and indicates which questions are addressed in this review. The Amphibians Workgroup focused on Questions 1 and 2 during this review cycle; Question 3 and its sub-questions are not addressed and are expected to be considered in future work.

AMPC Question	Question	Addressed in Review
Question 1	For each of the covered amphibian species, what is the distribution (including population diversity) within Oregon, and what factors (e.g., stream gradient, stream size, fish presence/absence, slope, aspect, temperature, seasonality, micro-habitat conditions) determine this distribution at a smaller spatial scale (e.g., watershed)?	Yes
Question 2	What is the population trend of the Columbia and Southern torrent salamanders over time on lands subject to the Forest Practices Act (FPA) rules? This question is informed by the following overarching biological goal as stated in the draft PFA HCP: “Forest practices that support the survival and recovery of the covered species by providing clean, cool, connected, and complex habitats.”	Yes
Question 3	The following sub-questions are informed by direction from the PFA Report to “…better understand how riparian and unstable slope protections of at least the current and proposed rules for private forestland impact persistence of populations.”	Not yet addressed
Question 3.1	How do rules for no-harvest RMAs affect Columbia and Southern torrent salamanders’ habitat? BGO from Draft PFA HCP: Goal 2: Shade and watershed processes controlling stream temperature provide cool water compatible with the needs of the covered species. Objective 2.2 – No-harvest RMAs maintain stream shade sufficient to support desired cool water temperatures for covered amphibians. The most recent version of the BGOs is in the Dec. 2022 draft HCP. The BGOs will be finalized within the HCP due Dec. 31, 2027. Private Forest Accord Report, p. 121.	Not yet addressed
Question 3.2	How do rules for Type N streams affect Columbia and Southern torrent salamanders’ habitat? BGO from Draft PFA HCP: Goal 3: Stream network connectivity satisfies freshwater habitat needs for covered species. Objective 3.3 – Timber harvest maintains stream-associated connectivity in riparian areas along non-fish streams sufficient to support covered amphibians.	Not yet addressed
Question 3.3	How do rules for steep/unstable slope protections affect Columbia and Southern torrent salamanders’ habitat? BGO from Draft PFA HCP: Goal 4: Riparian areas function to support complex habitats for the covered species. Objective 4.3 – Designated Debris Flow Traversal Areas function to deliver large wood to fish-bearing streams. Objective 4.4 – Forest practices maintain stream-associated wetlands and stream-adjacent seep and spring habitat for amphibians.	Not yet addressed