Effects of Forest Roads and Tree Removal In or Near Wetlands of the Pacific Northwest: A Literature Synthesis

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WASHINGTON STATE DEPARTMENT OF
Natural Resources
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Washington State Forest Practices Adaptive Management Program

The Washington State Forest Practices Board (FPB) has established an Adaptive Management Program (AMP) by rule in accordance with the Forests & Fish Report (FFR) and subsequent legislation. The purpose of this program is to:

> *Provide science-based recommendations and technical information to assist the FPB in determining if and when it is necessary or advisable to adjust rules and guidance for aquatic resources to achieve resource goals and objectives. The board may also use this program to adjust other rules and guidance.* (Forest Practices Rules, WAC 222-12-045(1)).

To provide the science needed to support adaptive management, the FPB established the Cooperative Monitoring, Evaluation and Research (CMER) committee as a participant in the program. The FPB empowered CMER to conduct research, effectiveness monitoring, and validation monitoring in accordance with WAC 222-12-045 and Board Manual Section 22.

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This technical report contains scientific information from research or monitoring studies that are designed to evaluate the effectiveness of the forest practices rules in achieving one or more of the Forest and Fish performance goals, resource objectives, and/or performance targets. The document was prepared for the Cooperative Monitoring, Evaluation and Research Committee (CMER) and was intended to inform and support the Forest Practices Adaptive Management program. The project is part of the Eastside Type F Riparian Effectiveness Program, and was conducted under the oversight of the Riparian Scientific Advisory Group (RSAG).

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1.0 Introduction

In the Pacific Northwest, hundreds of studies over the past three decades have focused on the effects of timber harvests and logging roads on the functions and species in *streams*. However, few studies have examined the effects on *wetlands*. Selected studies covering this topic in the Pacific Northwest were described by Cooke Scientific Services (2005) under contract to the Cooperative Monitoring Evaluation and Research Committee/Wetland Scientific Advisory Group (CMER/WETSAG). Elsewhere in North America, literature on the effects of forest practices on wetlands in the southern United States was reviewed in part by Shepard (1994), Aust (1994), Conner (1994), Sun et al. (2001), and others, whereas much of the literature on forestry impacts to wetlands in Canada was reviewed by Smith et al. (2007). The very limited literature on forestry impacts to lakes of the Pacific Northwest (mainly British Columbia) was compiled by Miller (1997).

Because studies of forestry effects on wetlands have been so infrequent in the Pacific Northwest, each section in this report drew heavily from studies of forestry impacts to streams and riparian zones. After assembly and synthesis, that information was extrapolated, mostly in the form of hypotheses (Appendix A), to the very different conditions known to be present in the region's wetlands. Inferences were based largely on the author's knowledge of wetland functions and decades of experience as a wetland scientist in this region. In a similar manner, information from pertinent studies of the region's wetlands was used to hypothesize the possible effects of forest practices on wetlands even when a particular study did not specifically address the separate effects of timber harvests, logging roads, or silvicultural chemicals. As such, this study provides an initial framework for considering how to examine the reality and perhaps the extent of those effects.

1.1 Scope of the Synthesis

In the context of wetlands, this report addresses the physical, chemical, and biological effects of diverse forest practices. These effects may be the direct or indirect result of tree removal (i.e., logging, timber harvest), roads and other infrastructure created in support of logging operations, or use of silvicultural chemicals. Where possible, the separate effects of these are distinguished, and with regard to both their magnitude and duration. Within the category of "tree removal," effects resulting from clear-cutting of trees are distinguished from effects of partial (selective) harvests, as available information allows. This report addresses not only the effects of cutting trees within forested wetlands (i.e., the effects of on-site harvests), but also the effects on wetlands -- of any type -- where timber is harvested in nearby uplands (i.e., the effect of off-site harvests). In many cases, the effects of both on-site and off-site harvests depend on wetland type, so distinctions among effects based on wetland type are noted when supported by available science (see section 1.3 for brief description of wetland classifications used). Effects of off-site harvests on wetlands also depend on the width and other characteristics of wetland ***buffers,*** also called streamside, riparian, or wetland management zones (SMZ's, RMZ's, WMZ's), filter strips, setbacks, or streamside/ riparian reserves. These are areas in which existing vegetation generally remains unharvested as one means of potentially reducing the potentially adverse effects of harvesting trees or other land uses on nearby water bodies. This report reviews the literature on the effectiveness of buffers as it may pertain to wetlands in forested areas of this region. Geographically, this report defines the Pacific Northwest (PNW) region as Oregon, Washington, British Columbia, and Southeast Alaska. To a lesser extent, studies from other western states and provinces along the U.S.-Canada border were considered, as well as (to a lesser degree) studies from more easterly states and provinces along the border. Those studies were accorded lower priority for review because of limited time and resources for this review, and because their flora and fauna differ significantly from that in the Pacific Northwest.

1.2 Role of the Synthesis

In this region, most natural resource agencies have adopted rules or recommendations regarding whether and under what circumstances (season, wetland sizes and types, geomorphic settings, types of buffers, equipment, harvest methods, etc.) timber harvests and road-building may occur in or near streams and/or wetlands. This report is intended to capture the best available science on its topic in this region. It is not intended to be a review of the technical adequacy of any specific regulation or rule in the State of Washington (WAC 222-16-035, -036, and others) or elsewhere. Information on some of the forest practices regulations in this region and compliance monitoring associated with those has been compiled and compared by Broadmeadow and Nisbet 2004, Michael 2004, Adams 2007, Council of Western State Govenors 2007, Tschaplinski and Pike 2010, and others.

1.3 Wetland Classes and Categories

As noted above, effects of forest practices on wetlands -- and consequently selected regulations that govern those practices -- depend partly on wetland type. That is because (1) wetlands differ with regard to their functions and the factors that determine those, and (2) wetlands appear to differ with regard to their sensitivity (resilience and resistance) to various components of forestry operations, although data from this region are limited. Many schemes have been devised for scientifically classifying or administratively categorizing wetlands in this region (see review: Adamus 2004). The three referred to in this report are those used most often by agencies in Washington or the Pacific Northwest:

1. State of Washington Forest Practices Code

The Code categorizes wetlands administratively as Forested Wetland, Bog, Type A, or Type B. Depending on category, they must be ¼ or ½ acre in extent in order to be subject to forest practices rules. *In brief*, **Forested** wetlands are any wetland or portion thereof that has (or would have, if the trees were mature) tree cover with a crown closure of 30% or more. **Bogs** are wetlands with organic soils and ground cover usually dominated by mosses, but sometimes by other woody or herbaceous plants named specifically in the Code. **Type A** wetlands are herbaceous wetlands on the fringes of ponds, lakes, or rivers that contain surface water for at least 7 consecutive days between April 1 and October 1. **Type B** wetlands are all other herbaceous wetlands.

2. Cowardin et al. (1979) Classification

This is a scientific classification used in the maps of the National Wetlands Inventory. It has many hierarchical levels that describe a wetland's major system (estuarine, lacustrine, riverine, palustrine), vegetation forms (e.g., emergent, scrub-shrub, forested), and water regime (e.g., permanently flooded, saturated, seasonally flooded). Under this classification, most wetlands categorized as Forested under the Washington Code (above) would be classified as Palustrine Forested or Palustrine Scrub-shrub. Most wetlands categorized as Bogs under the Washington Code would be classified as Palustrine with a Saturated water regime modifier under this classification, but not necessarily vice versa. Wetlands belonging to Type A or Type B belong to the Cowardin Emergent class, but cannot be assigned to a Cowardin system (e.g., riverine, Palustrine, Lacustrine, or Estuarine) without further information. All wetlands that the Cowardin classification describes as Lacustrine would be categorized as Type A wetlands by the Washington Code, but many other wetlands would be as well.

3. Hydrogeomorphic (HGM) Classification

This scientific classification (Brinson 1993, Table 1) emphasizes the predominant source of water to a wetland (groundwater, runoff, or direct precipitation), its direction (unidirectional or bidirectional), presence and direction of flow, and geomorphic setting (connected or isolated from other surface waters). Major classes are estuarine, riverine, lacustrine, depressional, slope, and flat. The first three of these do not correspond perfectly with their same-named classes in the Cowardin classification. Wetlands categorized as Forested under the Washington Code occur in all of these HGM classes (but are rare in the HGM estuarine class). Most wetlands categorized as Bogs under the Washington Code would be classified as flats or slope wetlands, or perhaps depressional. Nearly all wetlands that the HGM classification describes as lacustrine or riverine would be categorized as Type A wetlands by the Washington Code, but many other wetlands would be as well.

Because they are based on hydrology, vegetation, and water chemistry, none of these three classifications correlate directly with wetland use by fish and wildlife. Thus, if these classifications alone are used as the basis for regulations, they may be suboptimal for protecting the diversity of native fish and wildlife species. In such instances, the complementary use of additional classifications (e.g., stream typing, as defined in Washington State by WAC222-16, or hydrologic landscape classification – Wigington et al. 2013) may be helpful. Similarly, a random selection of research sites stratified only by HGM class and subregion will likely fail to optimally address the variability in wetland use among different wildlife and fish species.

Table 1. Definitions of hydrogeomorphic (HGM) classes of wetlands, as interpreted from Brinson (1993), Smith et al. (1995), and Adamus (2001)

Hydrogeomorphic Class	**Water Sources***	**Flow Direction**	**Examples**
Riverine	runoff> groundwater> precipitation	Unidirectional (channels) & bidirectional (floodplain)	Wetlands along streams, rivers, ditches with flowing water.
Depressional	runoff> groundwater> precipitation	Vertical (seepage)	Wetlands in ponds, potholes. Often in headwaters.
Mineral Soil Flats	precipitation > groundwater> runoff	Vertical (seepage)	Vernal pools. Usually lack natural outlets.
Organic Soil Flats	precipitation > groundwater> runoff	Vertical (seepage)	Bogs, some fens.
Slope	groundwater> runoff	Unidirectional, horizontal	Swales, stream headwaters.
Lacustrine Fringe	runoff> precipitation > groundwater	Bidirectional (seasonal overflow), horizontal	Wetlands along lakes (>20 acres)
Estuarine Fringe	ocean> runoff> groundwater	Bidirectional (from tide), horizontal	Salt marshes, tidal swamps.

* "runoff" is surface flow from catchment, "groundwater" is lateral or vertical subsurface flow. These rankings are not quantitative and in any given wetland, they may change seasonally and interannually.

1.4 Endpoints: Wetland Functions and Wetland Condition

The focus of this review is primarily on impacts to wetland ***functions*** because functions are the endpoints most commonly identified for protection in Washington wetland regulations. Functions are what wetlands potentially do, such as store water and other natural processes, regardless of how much those processes directly affect humans. However, because wetlands have hundreds of potential functions, judgment was exercised in selecting *which* functions (and their level of aggregation) were to be the focus of this review. An explicit decision was made to focus on the hydrologic (water regime), water quality (sediment, phosphorus, nitrate, carbon), microclimate-sustaining, and habitat-supporting (for wetland plants, invertebrates, amphibians, turtles, wetland birds and mammals, fish) functions. This was done largely because these are the functions, and the level of function aggregation, recognized by the Washington Department of Ecology and many other agencies when addressing wetland issues.

Another endpoint is wetland ***condition***. That describes a particular state, usually one defined by features that are specified by the investigator. Wetland condition is often used interchangeably with terms such as wetland health, integrity, quality, or intactness, all relative terms that carry considerable definitional ambiguity.

Despite popular perceptions, high-functioning wetlands are not always considered to be in good condition and wetlands believed to be in good condition are not always high-functioning. This is true for at least three reasons (De Leo and Levin 1997, Hruby 1999, McLaughlin and Cohen 2013) : (1) There exists no widely-accepted scientific definition of what wetland conditions should be, (2) There exists no widely-accepted scientific protocol for measuring wetland condition comprehensively (all taxa and processes, or using fewer taxa or processes known to correlate with all others), and (2) No single wetland, regardless of how intact or pristine it may be, can provide all functions at their highest potential level because many wetland functions operate naturally in opposing directions.

With regard to reasons #1 and #2, no scientific consensus exists as to which individual or combined processes (primary productivity? sedimentation rates?), biological group or groups (birds? plants? microbes? algae?), or metrics (species richness? stem density? tolerance indices?) most accurately represent a wetland's ecological condition Although some processes, taxonomic groups, and/or metrics often respond in a similar direction to pollution, habitat alteration, and other stressors (e.g., Rooney & Bayley

2012), in other cases the responses show little concordance. That may depend on which contaminant or type of habitat alteration (or their combination) is locally dominant, its intensity, compensatory factors such as natural resilience and resistance of some types of landscapes, the scale of measurement, and other factors difficult to predict beforehand. Thus, although some groups (notably plants) and metrics (percent cover of exotic species) are used to represent what the authors who study them believe is wetland ecological condition (typically assuming the absence of human influence is synonymous with the best ecological condition), no research has shown that PNW wetlands that support (for example) a rich variety of native plants also support a rich variety of fish, aquatic invertebrates, microbial communities and processes, waterbirds, etc. Amphibians, microbes or other taxa may respond to pollution and habitat alteration quite differently than plants, in some unknown proportion of the wetlands. It also is likely that their separate response to pollution and habitat alteration is indistinguishable, without using prohibitively large sample sizes, from their response to natural variation. An impoverished fish community, for example, might be the result of limited wetland connectivity with other waters rather than pollution, and reduced functional diversity of microbes could be the result of naturally cloudy or acidic waters. Many instances of "disturbance" to wetlands (if that can be defined objectively) exist that result in sustainable increases, not decreases, in levels of some native wetland species and functions.

With regard to the second point made at the beginning of this section – that no single wetland, regardless of how pristine it may be, can support *all* functions at a high level because many wetland functions operate naturally in opposing directions – this is supported by several analyses of field data (e.g., Hansson et al. 2005, Adamus et al. 2009, Acreman et al. 2011). For example, wetlands most effective for storing water (such as those with naturally large water level fluctuations) are not necessarily the most effective for functions such as supporting pollinating insects or habitat for wetland plants. Similarly, wetlands most effective for supporting fish are not necessarily the most effective for supporting amphibians and aquatic invertebrates (which are preyed upon by fish in diverse circumstances). Thus, it is misleading to describe a wetland as having "high function" or being "highly functional" without specifying the function or combination of functions to which one is referring.

In summary, although *generally* high levels of many wetland functions can be expected to correlate positively with *generally* high levels of ecological condition (intactness, integrity, health, etc.), a *causal* connection has never been proven and should not be automatically assumed. Any correlation will depend on how functions and indicators of condition are measured, the types of stressors to which particular wetlands are being exposed, spatial variation of natural factors within the landscape, and other influences.

1.5 Wetlands in Forests of the Pacific Northwest

Forested wetlands are likely the type of wetlands most often subject to on-site or off-site tree harvest. In 2004, a survey of the bankfull zone of headwater streams in 30 watersheds in the Washington Coast Range (Janisch et al. 2011) revealed an average of 2.3 wetlands per first-order channel. All were smaller than 0.1 ha and occurred mainly where (a) their contributing areas were north-facing, (b) surface water was perennial, and/or (c) channels contained large wood originating from adjacent riparian forest. About 40% of the channels surveyed were sourced by a channel-head wetland. Several wetlands were associated with streamside topographic depressions left by root balls of toppled trees. As they rapidly fill with water and fine sediments, those depressions may facilitate wetland formation. The authors commented:

> "Were our sample representative, every 1000 such catchments would support, on average, ~19 ha of wetlands not typically surveyed. Given that river miles of headwaters greatly exceed that of mainstem rivers, headwater wetland area could, in rugged topography, rival or exceed that of lowlands. Whether wetland area or stream area dominates headwater catchments is thus a key question."

Yet, nearly all studies of the hydrology of headwater catchments have focused only on channels, not on wetlands. Ecologically, the influence of these tiny wetlands could be significant. For example, headwater wetlands one-third of an acre or less (< 1,335 m^2) can increase the duration and magnitude of stream discharge as well as affect stream chemistry and fish access, particularly during periods of base flow (Morley et al. 2011).

Several publications have described in detail the plant community composition of wetlands in parts of the Pacific Northwest, in some instances organizing the information as a vegetation-based classification. Examples are Murray 2000, MacKenzie and Banner 2001, MacKenzie and Shaw 1999, MacKenzie and Moran 2004, Christy 2004, Crowe et al. 2004, Kovalchik and Clausnitzer 2004, Wells 2006, and Rocchio et al. 2012. What follows is a very general listing of some of the major plants found in forested wetlands in all or part of the PNW region. These descriptions set the stage for discussions of forestry impacts to wetlands later in the report.

Conifer **trees** occurring as dominants in the region's forested wetlands are, perhaps most frequently:
western hemlock (*Tsuga heterophylla*), western red-cedar (*Thuja plicata*), Sitka spruce (*Picea sitchensis*), Engelmann's spruce (*Picea engelmannii*), and (primarily in bogs) lodgepole pine (*Pinus contorta*). Often, the deciduous species occurring as dominants in the region's forested wetlands are willow (*Salix* spp.), alder (*Alnus* spp.), black

cottonwood (*Populus trichocarpa*), Oregon ash (*Fraxinus latifolia*), quaking aspen (*Populus tremuloides*), and paper birch (*Betula papyrifera*). Douglas-fir (*Pseudotsuga menziesii*), which dominates forests and timber harvesting in much of the western part of the region, is not a typical dominant in the region's wetlands. In the PNW, most forests next to streams and wetlands include more deciduous trees, fewer conifers (though sometimes with equal or higher basal area), a more open canopy, more berry-producing shrubs, and fewer evergreen shrubs than adjacent upland forest (Pabst and Spies 1999, Pearson and Manuwal 2001). Wetlands at lower elevations tend to be dominated by deciduous trees and shrubs to a greater degree than wetlands at higher elevations.

In western parts of the region, native **shrubs** that are dominant in forested wetlands are, perhaps most often: salmonberry (*Rubus spectabilis*), snowberry (*Symphoricarpos albus*), elderberry (*Sambucus racemosa*), red osier dogwood (*Cornus sericea*), hardhack (*Spirea douglasii*), black hawthorn (*Crataegus douglasii*), western crabapple (*Malus fusca*), willow (*Salix* spp.), and alder (*Alnus* spp.). In forested wetlands of Southeast Alaska, blueberries (*Vaccinium* spp.), several native berries (*Rubus* spp.), and devil's club (*Oplopanax horridum*) are frequent dominants.

Native herbaceous plants that often dominate the understory of forested wetlands in the western part of the Pacific Northwest commonly include: slough sedge (*Carex obnupta*), skunk cabbage (*Lysichitum americanum*), stinging nettle (*Urtica dioica*), touch-me-not (*Impatiens* spp.), and Pacific golden saxifrage (*Chrysosplenium glechomifolium*). In coastal British Columbia and Southeast Alaska, bluejoint reedgrass (*Calamagrostis canadensis*) and several moss and sedge species dominate many wetlands.

Several plants not originally native to the Pacific Northwest have invaded many of the region's forested wetlands. Non-native **shrubs and vines** that are perhaps the most widespread invaders of the region's forested wetlands include Himalayan blackberry (*Rubus armeniacus*), cut-leaf blackberry (*Rubus laciniatus*), climbing nightshade (*Solanum dulcamara*), and English ivy (*Hedera helix*). However, these species do not tolerate inundation or saturated soils for long and thus seldom become dominant throughout any forested wetland. In western Oregon and Washington, non-native **herbaceous** species that are perhaps the most widespread and dominating invaders of wetlands in forested landscapes include reed canary-grass (*Phalaris arundinacea*), Canada thistle (*Cirsium arvense*), creeping bentgrass (*Agrostis stolonifera*), creeping buttercup (*Ranunculus repens*), bird vetch (*Vicia cracca*), garden vetch (*Vicia sativa*), scented bedstraw (*Galium odoratum*), and giant knotweed (*Polygonum sachalinense*).

2.0 Literature Synthesis Methods

2.1 Identifying and Prioritizing Sources

A systematic literature synthesis was conducted, generally following the steps described by Pullin and Stewart (2006):

1. Formulate questions
2. Define and implement a literature search strategy
3. Index and prioritize the identified literature
4. Read and extract key information
5. Synthesize the information, partly by identifying connections among subtopics
6. Peer review

The first step – formulating the main questions – was completed by the WETSAG prior to project start-up. Those questions are shown in Table 2.

Table 2. Main questions to be addressed by this literature synthesis (from WETSAG)

<u>Water Regime Questions</u>

1. What are the potential effects from clear-cutting wetlands, and the magnitude and duration of those effects on the water regime of wetlands?

- What factors (For example: wetland soils, HGM type, timing of clear-cut, location of wetland in the watershed) affect the magnitude and duration of the hydrologic response?
- How do these hydrologic changes affect the opportunity and potential[2] of wetlands to maintain fish and amphibian habitat and productivity in a watershed?

2. What are the potential effects from partial harvesting of wetlands, and the magnitude and duration of those effects on the water regime of wetlands?

- What factors (For example: wetland soils, HGM type, timing of partial harvest, location of wetland in the watershed) affect the magnitude and duration of the hydrologic response?

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[2] Hruby, T. 2004. Washington State wetland rating system for western Washington – revised. Washington State Department of Ecology Publication # 04-06-025, pg. 32-35. The DOE rating system defines "Potential" as using structural wetland characteristics as indicators of the capability or "potential" of a wetland for performing a function. "Opportunity" is defined as the second part in characterizing the wetland function by characterizing to what degree a wetland's position in the landscape will allow it to perform a specific function. Opportunity and potential are both integral parts of the DOE rating system for rating wetland functions.

- How do these hydrologic changes affect the opportunity and potential of wetlands to maintain fish and amphibian habitat and productivity in a watershed?

3. What are the potential effects of road construction and maintenance activities in/or adjacent to wetlands, and the magnitude and duration of those effects on the water regime of wetlands?

What factors (For example: wetland soils, HGM type, timing of road construction, location of wetland in the watershed, location of road relative to wetland and other aquatic resources) affect the magnitude and duration of the hydrologic response?

- How do these hydrologic changes affect the opportunity and potential of wetlands to maintain fish and amphibian habitat and productivity in a watershed?

Water Quality Questions

4. What are the potential effects of clear-cutting wetlands, and the magnitude and duration of those effects on the water quality (temperature, suspended sediment, nutrient loads, pH, dissolved oxygen, toxicity) of wetlands?

- What factors (For example: wetland soils, HGM type, timing of clear-cut, location of wetland in the watershed) affect the magnitude and duration of the water quality response?
- How do these changes in water quality affect the opportunity and potential of wetlands to maintain fish and amphibian habitat and productivity in a watershed?

5. What are the potential effects of partial cutting, and the magnitude and duration of those effects on the water quality (temperature, suspended sediment, nutrient loads, pH, dissolved oxygen, toxicity) of wetlands?

- What factors (For example: wetland soils, HGM type, timing of partial harvest, location of wetland in the watershed) affect the magnitude and duration of the water quality response?
- How do these water quality changes affect the opportunity and potential of wetlands to maintain fish and amphibian habitat and productivity in a watershed?

6. What are the potential effects from road construction and maintenance activities in/or adjacent to wetlands, and the magnitude and duration of those effects on the water quality (temperature, suspended sediment, nutrient loads, pH, dissolved oxygen, toxicity) of wetlands?

- What factors (For example: wetland soils, HGM type, timing of road construction, location of wetland in the watershed, location of road relative to wetland and other aquatic resources) affect the magnitude and duration of the water quality response?
- How do these water quality changes affect the opportunity and potential of wetlands to maintain fish and amphibian habitat and productivity in a watershed?

Wetland Management Zone Question

7. What are the effects of leaving buffers around wetlands on maintaining wetland functions that sustain fish, amphibians and water quality?

Cumulative Effects Questions

8. What are the potential cumulative effects (spatial and temporal) of clear-cutting wetlands on watershed processes that support fish, amphibians and water quality?
9. What are the potential cumulative effects (spatial and temporal) of partial harvesting wetlands on watershed processes that support fish, amphibians and water quality?
10. What are the potential cumulative effects (spatial and temporal) of constructing and maintaining roads in/or adjacent to wetlands on watershed processes that support fish, amphibians and water quality?

<u>Parameters and Metrics</u>

11. What parameters and metrics can be used to assess, evaluate and quantify the effects of forest practices on the water regime and water quality of wetlands, and fish and amphibian productivity and use of wetlands?

Although those questions were essential to guiding the synthesis and all are addressed in this document, they were slightly re-organized in the presentation of this document. Specifically, clear-cutting and partial cuts were combined into "Tree Removal Effects" subsections because of the unfortunate paucity of studies that have distinguished their effects. Nonetheless, when information that distinguished the effects of these was available, it was highlighted in the narratives. Questions above that are related to "how changes affect the opportunity and potential of wetlands to maintain function ___" as well as the questions about cumulative effects are discussed (when information allows) mostly in subsections titled "Biological Effects of [type of forest practice]." The question on Wetland Buffer Management Zone effectiveness is treated separately within each major chapter (Water Regime, Water Quality, and Microclimate-Vegetation-Animals).

At the project outset, a database was created in Excel® for references that might be included in this synthesis. The database was selectively populated with citations of publications from the literature cited sections of several prior syntheses on related topics. The synthesis reports were searched manually for citations that appeared useful based on their titles, and included (for example): Adamus et al. (2001), Cooke Scientific Services (2005), and Pike et al. (2010).

Next, sets of keywords were identified in three categories: Wetlands, Forest Practices, and Geography. Over a dozen synonyms or closely related terms were included in the Wetlands category, including (for example) swamp, marsh, bog, fen, riparian, pond, off-channel, seep, amphibian, and waterfowl. Terms in the Forest Practices category included (for example): logging, forest road, clearcut, timber harvest, partial harvest, buffer, herbicide, and fragmentation. Geographic terms were applied that define the states, provinces, and major features (e.g., Cascades) of the Pacific Northwest, and secondarily other states and provinces near the U.S.-Canada border. Research on the effects of forest practices on wetlands in the southern United States was not reviewed because of the need to prioritize the literature reviewed, due to time and resource limitations, as well asdifferences in climate, topography, species, and wetland types of the PNW increase the level of uncertainty when extrapolating between these regions.. Likewise, research on forested wetlands in the more northerly boreal regions of Canada and Alaska, where artificial drainage of peat soils is often a prerequisite for timber harvest, was mostly excluded.

Initially, the automated keyword search tool that was used was the online bibliographic database, *Web of Science*®. It allows complex Boolean querying of its millions of listings that go back several decades, and provides abstracts from most of the journals it references. Several queries of *Web of Science*® using the chosen keywords resulted in a list of over 5000 papers and reports. By manually reviewing the title of each of these (and where necessary, reading the abstract), the principal investigator narrowed the list to about 800 citations. These were added to the Excel® database of citations selected previously from the prior syntheses (as described above) and duplicates were eliminated.

2.2 Indexing and Prioritizing the Literature

At this point, cell formulas were written in the Excel® database to search the title and abstract of each database citation for particular keyworded topics, one per database column, e.g., evaporation, macroinvertebrate, soil compaction, depressional, saturated. The cell formula places a "1" in any cell where the term specified in the column is found in the title or abstract of a citation (row). In this manner the database was indexed automatically. Using Excel's "sort" tool, the citations can be grouped by any topic or combination of topics, and re-sorted as desired. Additional manual review and indexing of the citations allows them now to be sorted by wetland type, publication year, and geography.

Next, each of the 800+ citations was categorized as A, B, or C to reflect the priority for obtaining and reading it. Higher priority was assigned to publications that described original PNW research involving both wetlands and forest practices, or which, from their titles or abstracts, appeared to be most relevant to the objectives of this project. Lower priority was assigned to literature reviews and "white papers" (not original research), to publications not in peer-reviewed journals, and to publications about wetlands or forest practices but not both. A mostly successful effort was then made to obtain an electronic or hard copy of the full publication referenced by each A-priority citation and most of the B-priority citations. When a publication could not be obtained, an attempt was made to at least obtain its abstract, and all obtained abstracts were read. Finally, nearly all the A-priority publications, most of the B-priority citations, and some of the C-priority citations were read. Relevant information from each was incorporated directly while writing this report, rather than creating an intermediate step wherein the relevant information was first extracted to a separate file before being written up.

In all, 669 publications (171 priority A, 471 priority B, 27 priority C) were read in whole or part and included in the database. Information from a few additional publications recommended by reviewers was incorporated into the final version of this report and its Literature Cited section, but was not catalogued in the database. Of the 669 publications catalogued, 493 (74%) are primary literature (based on original field data or analyses) and the rest are literature syntheses and opinion documents. Nearly half were published since 2002 and 14% were published since 2009. Of primary literature sources, 284 (57%) mention wetlands, 287 (58%) mention forest practices, 240 (49%) pertain to studies in the Pacific Northwest, and 85 (17%) have all three elements, i.e., are from the PNW and mention both wetlands and forest practices. It is important to understand that: (a) simple mention of wetlands and forest practices in the same publication does not mean that effects of the practices on wetlands were studied, and (b) many studies of stream riparian areas were assumed to be studies of river-fringe wetlands, when in fact they may not have included wetlands at all, because not all (or even most) riparian areas are jurisdictional wetlands. Thus, the actual number of studies of forest practices in twetlands (as defined in this review) in the PNW is far less than the 85 indicated. Without better description of the study site conditions, the true number is impossible to determine. Tables 3-6 contain tabulations of the numbers of primary literature publications from the PNW that were reviewed, by topic.

Table 3. Number of reviewed PNW field studies of sites known or assumed to be wetlands, by HGM class.

Depressional	1
Lake Fringe	1
River Fringe	83
Slope	4
Tidal	1
Undeterminable HGM class	44

Table 4. Number of reviewed PNW field studies of sites known or assumed to be wetlands, by Washington Forest Practices Code type.

Bog	2
Forested	42
Type A or B (indeterminable)	90

Table 5. Number of reviewed PNW field studies of sites known or assumed to be wetlands, by topic

Nearly all were studies of stream riparian sites, not necessarily jurisdictional wetlands.

	#	%
Timber Harvest	78	58%
Forest Roads	24	18%
Forest Chemical Applications	6	4%
buffers	27	20%
hydrology	28	21%
ground water	11	8%
biogeochemistry	50	37%
temperature	23	17%
microclimate	10	7%
fragmentation	25	19%
algae	7	5%
invertebrates	17	13%
fish	24	18%
amphibs	24	18%
birds	13	10%
mammals	10	7%

Table 6. Number of reviewed publications by publication year and type

Publication Year	Field Studies	Review Papers	Total
1967	1	0	1
1968	1	0	1
1970	1	1	2
1971	0	1	1
1972	3	0	3
1973	3	0	3
1975	4	0	4
1977	2	1	3
1978	2	0	2
1979	1	3	4
1980	2	4	6
1981	5	1	6
1982	2	2	4
1983	2	0	2
1984	3	0	3
1985	5	0	5
1986	6	0	6
1987	6	1	7

Publication Year	Field Studies	Review Papers	Total
1988	8	5	13
1989	5	5	10
1990	6	2	8
1991	9	0	9
1992	4	3	7
1993	11	4	15
1994	9	7	16
1995	9	9	18
1996	16	0	16
1997	26	3	29
1998	17	4	21
1999	11	6	17
2000	30	6	36
2001	24	7	31
2002	22	5	27
2003	30	8	38
2004	17	11	28
2005	28	13	41
2006	39	7	46
2007	35	10	45
2008	26	13	39
2009	29	17	46
2010	19	15	34
2011	12	1	13
2012	2	1	3

2.3 Understanding the Limitations of the Best Available Science

Uncertainties in understanding the effects of logging operations on wetlands have arisen not only from a lack of such studies in the PNW, but also from past difficulties in designing and conducting studies of streams and watersheds that would have resulted in more definitive conclusions that could be applied to wetlands. Past investigations have used a wide variety of sampling and analytical methods which, when combined with apparent natural variation across time and space in hydrologic interactions, presents a challenge to making reliable, unified inferences about the relative influences of logging and natural processes on responses of wetlands and their functions. Ideally, most research should feature manipulative forest management experiments with pre- and post-treatment data, random assignment of replicates to treatments, and analyses over multiple spatial and temporal scales. In contrast, nearly all research which has attempted to relate effects of forest management to hydrologic, water quality, and biological impacts to streams and wetlands in the PNW has been of short duration, with some studies only evaluating post-harvest conditions. Even when pre- and post-harvest conditions are compared, between-year differences in the amount and timing of precipitation and temperatures between years can confound inferences one might make from the data, most notably if unharvested reference (control) sites were not part of the study. Likewise, even when harvested vs. non-harvested landscapes are compared, differences in soils, topography, vegetation, specific harvest practices and their configuration, and other factors can limit inferences that might otherwise be made. Moreover, many studies have occurred in landscapes that were harvested previously, and for which the precise histories of harvest may be ambiguous or unknown, which limits effective comparisons between current and historical distributions of plants, amphibians, and other organisms. Too often, biological studies have considered only the presence or absence of a species, rather than analyzing its degree of dependence on wetlands and upland buffers. Few studies have measured the reproductive success and long-term sustainability of populations of species, rather than abundance or density. Even the studies with pre- and post-treatment data can be affected by time lags. Effects on groundwater quantity and quality may occur distantly in space and time. Similarly, the longevity of selected wetland species may result in treatment effects not being manifest for several years. Few attempts have been made to measure the adaptability and resilience of individual members of a species to potentially harmful impacts of logging. Also, forest practices in the PNW have changed dramatically over the last two decades, and information about responses to current management rules is very limited.

When reviewing the literature for this report, the above limitations were taken into consideration generally but not specifically. That is, it was impractical to record and then judge the experimental design and field methods behind every one of the 600+ publications that were reviewed for this report, or the extent to which their findings can accurately be extrapolated to all wetlands and landscapes in the region. Nonetheless, it is believed that this report's inferences that were categorized as being the most certain (see Summary, section 4.0) are adequately supported by the aggregate of published scientific literature.

3.0 Forest Practices Effects on Wetlands

3.1 Effects on Water Regimes of Wetlands

3.1.1 Tree Removal Effects on Water Regimes of Wetlands

The existence of wetlands, which as a group are intermediate between uplands and deeper water, is often hydrologically precarious. Small changes in the amount and timing of groundwater, surface water, and precipitation that reach a wetland can determine whether (and how rapidly) the wetland transitions to a different ecosystem type or remains a wetland, and if the latter, whether these hydrological changes significantly alter its flora, fauna, and functions in ways that are judged as negative or positive. By a host of mechanisms, natural disturbances (fire, insect defoliation, landslides, windthrow) can just as easily trigger hydrologic changes on the land surface, and conversely, can be caused or amplified hydrologic changes (e.g., increased slope failure from changes in water level, snowfall accumulation, and rainfall interception that follow tree loss). Such changes from natural disturbances – like those associated with logging -- can create or eliminate wetlands or alter the functions of existing ones (Geertsema and Pojar 2007). However, when tree harvest is added to the suite of natural phenomena already disturbing wetlands, especially when both coincide in space and time, then natural disturbances are sometimes amplified. Effects compiled across all harvested watersheds in a region can be additive or they may be cumulative, meaning their sum over time and space may be greater than additive, in either a beneficial or detrimental manner (Preston and Bedford 1988). The degree to which effects may be cumulative is often determined by the degree to which individual impacts are staggered in time or are distributed sequentially in space, e.g., harvest which advances progressively uphill over time in a watershed, as opposed to occurring at nearly-random locations throughout (Ziemer et al. 1991, Beschta et al. 1995, Elliot et al. 2010).

A key factor that determines wetland type and function is *water regime*. Water regime pertains to the depth, duration (hydroperiod), frequency, diurnal fluctuation, and seasonal timing of groundwater and surface water. Water regime can be described by probability of an event occurring (e.g., 10-year flood probability or recurrence interval) and by standard statistics (e.g., mean, minimum, maximum flow) for specified time periods (e.g., daily, spawning season, annual). A large suite of variables – not just water yield, peak flow, and base flow -- have been used as “indicators” to describe hydrologic change in watersheds, streams, and rivers (Konrad et al. 2005, Poff et al. 2006, 2010, Poff 2009, Poff and Zimmerman 2010, Gao et al. 2009, Merritt et al. 2010). A similarly large number could be used to characterize changes in wetlands. Determining which variables significantly influence the biology, chemistry, and physical environment in wetlands remains a challenge. In general terms, some indicator variables that apply to estimating the hydrologic effects of forest practices on wetlands include:

- volume of water inputting to wetland (i.e., water yield of contributing area) and its timing
- peak water level or flow within the wetland: magnitude (depth or rate) and timing
- minimum water level or flow: magnitude (depth or rate) and timing

- percentage of days annually with surface water or measurable flow (both continuous and total)
- fluctuation (variance) in water level or flow: daily or annual
- percent of wetland water budget derived from groundwater vs. surface runoff vs. direct precipitation (and snow vs. rain)

Wetlands are dynamic, but the natural, actual, or desired range of these variables among the region's wetlands is unknown. Small isolated headwater wetlands are perhaps most at risk from hydrologic changes occurring in their contributing areas (catchments) because their hydrologic inputs are usually the least. In glaciated landscapes, some wetlands that comprise only one-third of their catchment area can produce 50-70% of the annual streamflow, because wetlands often occur where groundwater intercepts the land surface (Verry and Kolka 2003).

Many but not all studies have shown that removal of trees near a stream or in a wetland causes a mean annual rise in the local water table (see compilations and reviews of the vast literature on this, most notably by Stednick 1996 and 2008, Miller et al. 1997, Scherer and Pike 2003, Moore and Wondzell 2005, Guillemette et al. 2005, Brown et al. 2005, National Research Council 2008, Grant et al. 2008, Mallik and Teichert 2009, Smerdon et al. 2009, Troendle et al. 2010, and Winkler et al. 2010). As regeneration occurs in cutover areas, the previous rates and amounts of water transfer between uplands and wetlands return. This usually begins within 3-7 years post-harvest (Beschta 2002) -- less if the area has not been clearcut (Thomas and Megahan 1998). Hydrologic recovery to pre-harvest conditions takes 10 to 20 years in some coastal watersheds but may take many decades longer in mountainous, snow-dominated catchments (Whitaker et al. 2002, Moore and Wondzell 2005). Recovery is measurably delayed if the logged area is repeatedly treated with herbicide to control weeds that compete with replanted trees (Hornbeck et al. 1993).

The probability of a harvest operation having an effect on a wetland's water regime is greatest if trees are removed directly from a wetland or, if removed from outside the wetland, the removal occurs close to and upslope from the wetland. Impacts are also greater if the size of the harvested patch is large relative to the size of that wetland's catchment. Several other factors influence the degree to which tree removal causes water tables to rise. Especially on windy south-facing forest edges during the summer, tree roots can transfer large amounts of soil moisture to foliage and then to the atmosphere via transpiration and evaporation (Keim and Skaugset 2003). This effectively removes some of the water before it can reach wetlands and streams. Trees also intercept significant volumes of rain and especially snow, allowing some of that retained water to evaporate before it can reach wetlands and streams located farther downslope (Troendle and King 1987, Winkler et al. 2005). Thus, when trees are removed from within or above a wetland, that potential source of liquid water becomes available, the water table often rises, and the wetland may receive more water. This has been suggested by the data from many studies of streams and watersheds in the Pacific Northwest, such as those by Hetherington (1982, 1987), Jones and Grant (1996), Troendle and Reuss (1997), Thomas and Megahan (1998), Beschta et al. (2000), Hudson (2001), McFarlane (2001), and MacDonald et al. (2003). If resulting increases in peak flows are great, the morphology of channels can be affected (Grant et al. 2008). This can create, expand, or shrink riverine wetlands. Depending on the soils and topography, the slashburning and soil compaction components of some harvest operations provide additional surface runoff to wetlands, at least during a few years post-harvest (Lamontagne et al. 2000). In addition, in snow-affected areas, clearcuts have sometimes been shown to cause greater runoff during rain-on-snow events (Berris and Harr 1987) and earlier peaking of streamflow (or wetland water levels). This could cause shifts in aquatic species composition.

On the other hand, harvest might measurably *reduce* runoff to streams and wetlands in some parts of the Pacific Northwest during low runoff periods, partly by temporarily eliminating trees that otherwise contribute water by intercepting fog (Harr 1982, 1983). During the autumn, streams in clearcut watersheds in the PNW tend to have lower flows than in uncut watersheds (Harr et al. 1975). Also, cutting or windthrow of trees in or near wetlands can increase open-water evaporation sufficiently to reduce water persistence in late summer (Petrone et al. 2007), especially in larger wetlands and/or in drier parts of the PNW. In wetlands, this can have potentially detrimental effects on dissolved oxygen and fish access, as well as affect the maturation rate of larval amphibians. In at least one instance, a reduction in summertime stream flows was alleged to be due not only to changes in the amount of riparian vegetation, but from a harvest-caused shift from conifers to deciduous species (Hicks et al. 1991). However, in some situations deciduous trees may trap more snow (Helvey 1971) and after autumn leaf-fall, they would not continue to lose water via transpiration as evergreen trees would . In any case, where tree removal exceeds approximately 14% of a catchment, this has been shown to temporarily increase daily and/or seasonal *fluctuations* in the water levels of small water bodies located downslope in at least one instance (Taylor 1993).

Compared to an equivalently sized area that is clearcut, do patch cuts and thinning generate less new runoff? Unfortunately, the number of studies of partial cuts or thinnings is far fewer than the number of studies of clearcuts (Troendle et al. 2010). However, a wetland study in northern Minnesota (Kolka et al. 2011) may provide clues. Treatments included an uncut control (i.e., the upland and buffer were uncut) and three treatments in which the upland was clearcut but the buffer was either uncut, partially harvested, or clearcut. Water levels in the wetland rose following all treatments, with the largest increase in the wetland with no buffer and a clearcut located upslope. Differences among treatments were greatest during the first year post-harvest. By the fifth year after harvest the water levels in the treatment wetlands were not significantly different from the control. A study in Quebec (Pothier et al. 2003) similarly found that when 0, 40, 50, 60, and 100% of the tree basal area surrounding a conifer-forested wetland was removed, the degree of water table rise during the first post-harvest year was proportional to the amount of tree area removed. The water table returned to its previous level within 5 years post-harvest. In contrast to these two investigations, a study of two boreal lakes -- one with extensive logging in the watershed and along the shoreline and the other with moderate logging and a shoreline buffer -- found no measurable difference in water reaching the lakes after their watersheds were logged (Steedman 2000). The larger volume of the lake might have rendered immeasurable any water table changes that occurred.

In general, about 20% of the tree basal area above a stream must be removed before a statistically significant change in mean annual flow can be detected (Hibbert 1967, Bosch and Hewlett 1982, Stednick 1996). From a review of 50 studies globally, Guillemette et al. (2005) recommended that "logging should not cover more than 50% of a watershed, to minimize the occurrence of peak flow increases above 50%, which are deemed to affect stream morphology significantly." Peak flow data were from within 5 years post-harvest. The studies were analyzed if they met these criteria: (a) results were from paired watershed studies with a calibration period, (b) one result per watershed per treatment was used, with two exceptions, (c) peak flow changes at bankfull discharge (recurrence interval 1–2 years) could be evaluated and (d) the statistical significance of changes had been reported. The extent of wetlands, if any, in the study watersheds was not reported. Based on a detailed statistical analysis of many decades of data from western and eastern North America, Jones and Post (2004) determined that in the 5 post-harvest years, water yield increases 6-8 mm at conifer forest sites and 2-3 mm at deciduous forest sites. In conifer forest watersheds the increased water yields can persist for up to 35 years.

All before-and-after comparisons of harvesting operations face the challenge of statistically separating harvest and/or road-building effects from annual variation in the amount, form, and pattern of temperature and precipitation (Alila et al. 2009). Likewise all paired-watershed and multi-watershed comparisons face the challenge of statistically separating harvest and/or road-building effects from effects of different soil types (texture and depth to bedrock), topography, aspect, elevation, and other factors -- some unquantifiable -- that are potentially important in creating functionally meaningful differences among the studied watersheds. At a fine scale, the spatial and temporal variability of soil water storage capacity -- as influenced by evaporation, precipitation, and groundwater flow direction and amount -- complicates interpretation of forest harvesting studies (Devito et al. 2005). Despite these difficulties, results from most independent studies point in the same direction; this leads credence to the patterns the results suggest as a whole.

3.1.2 Logging Road Effects on Water Regimes of Wetlands

Construction of new roads accompanies many timber harvests. Maintaining existing forest roads in good condition is also important both for economic (timber management, fire control, recreation) and water quality protective reasons. Depending on how roads are designed, constructed, and maintained, the effects of roads on wetlands and watershed hydrology can be undetectable or significant, and they can be short-term or long-term (Schuldiner et al. 1979). Roads can change the volume and/or rate of runoff, its timing, and the proportion of precipitation that infiltrates and becomes groundwater rather than runoff. These effects can rival or exceed those of the harvests themselves. Road-diverted flow paths often directly or indirectly lead runoff into wetlands, streams, or onto downhill slopes. Roads in lowlands, especially older roads that were built in narrow valley bottoms, sometimes directly impinge on channels and wetlands. Road fills can increase the channel gradient, scour new areas, and reduce opportunity for overbank flow, thus reducing wetland area or the flooding frequencies needed to maintain some wetland functions. Road crossings of streams can induce channel changes immediately downstream of the crossing due to the effects of focusing water through a narrow culvert or other constriction narrower than in an unaltered stream.

Old roads can sometimes be relocated to reduce their impacts. Hydraulic excavators, which can excavate and place materials more precisely, have widely replaced bulldozers that formerly were used for road construction on mountain slopes. Roads are now designed to minimize cut and fill volume by constructing no wider than necessary and by fitting as closely as possible to natural topography (NCASI Forest Watershed Task Group 2003).

Runoff from roads generally follows one of four pathways: infiltration back into the hillslope below the road with no delivery to streams; direct delivery at channel crossings; direct delivery through gullies formed below relief drains; or indirect delivery via overland flow below the road. Direct delivery at channel crossings is the most common and most rapid form of delivery, and occurs where roadside ditches and/or road tread runoff are directed to the stream crossing structure, whether it is a culvert, bridge, or ford. Delivery at stream crossings is controlled partly by the spacing of relief drains; i.e., if relief drains are located only a short distance from the actual crossing, less road surface area will deliver water directly to the crossing.

Components of forest roads include cutslope, tread, fillslope, and any additional widening for ditches, berms, or other artificial surfaces that are part of the road right-of-way (NCASI Forest Watershed Task Group 2003). Most roads constructed for timber harvests in the Pacific Northwest are unpaved, often with a gravel surface. Many are closed during at least part of the year, usually during the wettest time. There are also many inactive forest roads that were closed to traffic immediately after completion of log hauling and/or silvicultural activities such as slash disposal, thinning, and tree planting. In many of these, measures have been taken to re-establish vegetation on the road surfaces and to minimize chronic erosion and future washouts.

Results from dozens of studies on the effects of forest roads on runoff and stream flow in the Pacific Northwest have been analyzed and critiqued many times, including reviews by Reiter and Beschta 1995, Wemple et al. 1996, Austin 1999, Gucinski et al. 2001, Scherer and Pike 2003, NCASI Forest Watershed Task Group 2003, Coe 2004, and Winkler et al. 2010. None are specific to wetlands because too few wetland studies exist to review. Relatively few of the dozens of studies of harvest and watershed hydrology have attempted to separate the effects of timber harvests from effects of logging roads and skidder trails associated with the logging.

The hydrologic effects of new roads are attributable to the following processes:
- slowing and occasional impounding of runoff and channel flow,
- connecting, by means of excavated roadside ditches, of existing natural drainageways that run perpendicular to the road,
- excavating into slopes and subsurface water flow paths, which causes more water to flow on the land surface, and
- removing vegetation, just as logging does, with consequent changes in water table height.

Essentially, roads can increase peak stream flows by replacing subsurface flow paths with surface flow paths, doing so through capture of subsurface water in road cuts and by reducing the rate of infiltration into compacted surfaces. In Quebec, Guillemette et al. (2005) attributed heightened post-harvest peak flows to the fact that a logging operation had connected skid trails and road ditches with branches of the stream in that watershed. They reported that maximum peak flow increased by 63% when harvesting and associated roads reached 61% of the watershed area. During the five-year period after the watershed had been 85% harvested, the maximum increase in bankfull flow was 57%. Previously, patch cutting 31% of a 394 hectare basin using chain saws (no skidder trails or roads) did not significantly modify rainfall generated peak flows and storm flows (Plamondon et al. 1998, Plamondon and Ouellet 1980). Small, wet, steep, headwater areas may be the most hydrologically-sensitive areas with regard to both timber harvests and road building (Moore and Wondzell 2005, Smerdon et al. 2009).

However, as is true of timber harvest effects, considerable variability exists in responses of water tables and stream flow to new forest roads (Wemple et al. 1996, Smerdon et al. 2009). One study (Bowling and Lettenmaier 1997) found that peak flows increased at a larger watershed scale only if the road construction occurred in headwaters of the larger watershed. Road construction near or below the centroid of a large basin can desynchronize the runoff hydrograph at the downstream location, potentially leading to *decreased* peak flows. Such de-synchronization effects may be responsible for the erratic results from field studies of the effects of road construction on peak flows in experimental watersheds, some of which show peak flows increasing, some decreasing, and others showing no statistical change (NCASI Forest Watershed Task Group 2003). Similarly, wetlands themselves have been shown to have varying effects on flood volume and stream low flows, and simple classifications (such as headwater vs. mainstem wetlands) have failed to predict these hydrologic conditions (Bullock and Acreman 2003).

Describing the variable effects of roads at a local scale, the NCASI Forest Watershed Task Group (2003) noted:

> Road cut interception of subsurface flow can be very erratic; it is highly dependent on specific site conditions, and can also be affected by harvest on hillslopes above the road. For example, a road in the Pine Creek watershed in the Idaho Batholith intercepted 8.4 inches of subsurface flow from watershed area above the road, which represented about seven times the amount of direct runoff from the road surface (Megahan 1972). In the Lochsa River drainage in north central Idaho, a road located several hundred yards downslope of the drainage divide intercepted on the order of 7 to 8 acre-feet of subsurface flow per day per mile of road and on the order of 5 to 6 acre-feet of overland flow per day per mile of road during the peak snowmelt season in early June (Burroughs et al. 1972). Conversely, a road in the Olympic Mountains of Washington intercepted no subsurface flow; there was no cutbank seepage and no base flow in road ditches (Reid and Dunne 1984). In central Idaho, Megahan (1972) found that one section of road intercepted more than three times as much water per unit length of road as another road section.

The same authors observed the following at a watershed scale:

> It has been hypothesized that increases in peak flows from roads may be due to increases in "drainage density" stemming from connectivity between the road system and the stream network at channel crossings and through gullies below relief drains (Montgomery 1994; Wemple et al. 1996; Jones et al. 2000). Although various studies have suggested increases in "drainage density" of 23 to 60% due to roads (Bowling and Lettenmaier 1997; Montgomery1994; Wemple et al. 1996), road mileage that drains to streams is not the same as an increase in the

length of streams within the watershed (Jones et al. 2000). Surface water instream develops as subsurface flow through the soil and geologic mantle, accumulating to the point of saturation and causing channelized surface flow. Roads also intercept subsurface flow, but interception may be partial, and there is a compensating effect in that runoff from intercepted subsurface flow is not supplied to the slope below the road, so storm runoff from the portion of the watershed below the road is reduced (Bowling and Lettenmaier 1997). Furthermore, if drainage density did actually increase because of road connectivity, relationships developed for natural drainages linking flood peaks and low flows to drainage density should be suitable for predicting the flow changes that occur from increased "drainage density" from road construction. However, studies of road construction effects on experimental watersheds show no such effects (Harr et al. 1975; Rothacher 1973). Jones et al. (2000) suggested that indices based on the number of road-stream crossings, particularly at mid-slope locations, may be more reflective of potential road effects on watershed processes than road density alone.

Arguments are made that runoff from intercepted subsurface flow occurs rapidly as concentrated overland flow, and therefore tends to increase peak flows (Jones and Grant 1996). This effect may or may not occur, because subsurface flow often occurs in macropores, where flow velocities can exceed that of overland flow. In such situations, intercepting overland flow will reduce peak flows (Bowling and Lettenmaier 1997). In fact, Cheng et al. (1975) reported that compaction associated with logging in a coastal British Columbia watershed *reduced* peak flow magnitude and time to peak because large subsurface channels which normally transported stormwater were crushed (although no surface flow resulted), forcing storm flow through the soil matrix at reduced velocity.

In some of the earliest work on road effects on peak flows, Rothacher (1970, 1973) did not detect any significant increases in mean peak flows or annual water yields due to roads that occupied 8% of a 250 acre western Oregon watershed. In the Oregon Coast Range, Harr et al. (1975) reported that roads increased mean annual peak flow by 19% in a watershed with 12% of area inroads, but that watersheds with 3 to 5% of area in roads showed no significant increases in mean annual peak flows. Ziemer (1981) and Wright et al. (1990) found no significant increases in peak flows for a 1047 acre watershed in coastal northern California with 5% of the area in roads and 8% in tractor skid trails. In seven small forested headwater basins in northern Idaho (ranging in size from 69 to 365 acres) with 1.8 to 4.3% of area in roads, five watersheds experienced no significant increases in mean annual peak flows, peak flows decreased in one of the seven basins, and flow during spring snowmelt increased 30% in one watershed with amid-slope road occupying 3.9% of the watershed area (King and Tennyson 1984).

In western Oregon, Jones and Grant (1996) concluded that 6% of the area occupied by roads in a small (250 acre) watershed caused a (non-significant) 20% increase in mean annual peak flows. They also argued that roads and clearcutting in large watersheds (15,000 to 150,000 acre) have caused increases in peak flows in larger basins. They attributed much of the effect to altered flow paths caused by roads. However, using a different statistical methodology with the same data, Thomas and Megahan (1998) found significant increases in peak flows in the small watershed that was roaded and patch-cut, but only in the first 10-year period following harvest, which implied that the effect was due to harvest, with little effect from roads, as road effects should be more persistent. They found increases in peak flows of 15 to 40%, but percentage increases in peak flows decreased with increasing return interval, with no increases detectable for flows greater than the 2-year flow, which is generally considered the flow level where effects on stream

> channels begin to occur. In another re-analysis of the data set used by Jones and Grant (1996), Beschta et al. (2000) essentially agreed with the conclusions of Thomas and Megahan (1998), but concluded that "peak flow increases for 0.4- to 5-year return interval events have occurred on small watersheds as a result of clearcut silviculture and the accompanying effects of roads and broadcast burning as practiced in the mid-1960s." They concluded that peak flow increases in the small watershed were not detectable for flows of 5-year return interval or greater, and they further concluded that the analysis did not support Jones and Grant's conclusion that peak flows were increased in larger basins.

Another complicating factor is that much of the existing road system was designed and built before present standards were in place. Thus, an accurate assessment of effects at a watershed scale must consider the proportion of old roads to the newer roads that incorporate improved engineering design (Gucinski et al. 2000), but few published studies have done that.

3.1.3 Buffer Effectiveness for Maintaining Wetland Water Regimes

Moderation of stream flow, wetland water levels, or water table heights are not reasons usually given for requiring vegetated buffers along streams or around wetlands. With the possible exception of very confined headwater locations, buffers of only a few dozen meters width seem unlikely to affect the tendency of larger forestry operations to change the amount or timing of runoff. Nonetheless, forested buffers – like any forest with trees that transpire water – under some situations may partly offset runoff volume that has increased as a result of logging and logging roads farther upslope. Transpiration rates and volumes are commonly greater in buffers because of their riparian location and regularly replenished soil moisture, than in equally-sized vegetated areas beyond the riparian zone (Moore and Wondzell 2005). Transpiration rates decrease as buffer width increases, and may become negligible in woodlands larger than 100 ha (Herbst et al. 2007). At one site in Oregon, transpiration occurring primarily in riparian buffers (which covered only 0.1 to 0.3% of a catchment) was estimated to account for daily loss of 1-6% of summer streamflow (Bond et al. 2002).

However, it cannot be assumed automatically that more forest cover (i.e., wider vegetated buffer) means more water loss. Much depends on soil moisture levels. Those in turn are influenced by weather, soil type, proximity to water bodies and other forest edges, wind, aspect, shade, as well as tree species, age, rooting depth and mass, groundwater flow rates and direction, and other factors (Bladon et al. 2006, Moore and Heilman 2011).

Headwater wetlands themselves can sometimes be considered "buffers" in terms of their ability, at a watershed scale, to moderate flow extremes in downstream areas. This is especially likely to be the case where most of the wetlands are hydrologically isolated for most of the year and flood only seasonally (Todd et al. 2006). They perform this function by providing a confined vertical space for storing runoff, by evaporating water from unvegetated surfaces, transpiring water where vegetated, and together de-synchronizing runoff as it moves downslope through a watershed. However, the degree to which any given wetland may do this likely depends largely on local groundwater conditions which seldom can be predicted without significant expense.

3.1.4 Biological Effects of Altered Water Regimes

As Brown et al. (2005) note, some aquatic resources may benefit from increased water yield that temporarily results from tree removal, but others may not. Whether the effects are beneficial or harmful depends on seasonal timing of the increase (particularly its coincidence with periods of very hot or cold temperature) as well as its magnitude and duration.

For over a century foresters have noted the proliferation of new "wet areas" following harvest operations, and this sometimes has been such an economic concern (because it retards growth of commercially important timber) that strategies have been devised to speed the drying process by planting seedlings of tree species with the greatest evapotranspiration rates (e.g., Landhäusser et al. 2003). In Quebec, higher post-harvest water levels in wetlands are considered to be such an economic nuisance that ditches are dug in or around affected wetlands to partially offset the effect (Marcotte et al. 2010). If topography and soils are suitable, water table rises in flat non-wetland areas can create anaerobic soil conditions long enough to support plant assemblages that define wetlands, where none existed prior to tree removal. This appeared to be the case in a headwater wetland in the Washington Coast Range measured by Janisch et al. (2011) before and after timber was harvested by clearcutting its catchment. Four years after its catchment was clearcut, in one very small wetland they observed (1) an increase in the spatial extent from 3.6 m^2 to 9.5 m^2; (2) an increase in vegetation mass and a conversion from sparse skunk cabbage to nearly pure, dense bulrush; and (3) development of strong indicators of reducing conditions in the wetland soil. They noted that other post-logging changes they observed 2004–2009 suggested that small headwater wetlands are highly sensitive to hydrologic disturbance and may develop or fade at a time scale of several years. This may reflect short, emergent groundwater paths, meaning that hydrological effects from disturbances, such as result from timber harvesting of upslope areas in small catchments, occur rapidly.

Germination of woody plants may be reduced in previously "dry" wetlands (those chronically lacking surface water) due to higher post-harvest water levels, i.e., greater frequency and duration of sediment saturation or flooding (Croke and Hairsine 2006) as described further in section 3.2.2.4.

On most active forest roads, ditches are cleaned periodically to insure that water does not erode the ditch or the road surface. However, many road ditches comprise dispersal habitat or even oviposition habitat for amphibians. Cleaning could potentially resuspend sediment and remove vegetation where eggs are deposited. From their field observations of very small headwater wetlands in the Washington Coast Range, Janisch et al. (2011) postulated that small headwater wetlands, regardless of their origin,may be quite important to several amphibians -- a viewpoint also expressed by other herpetologists (Olson et al. 2007).

Culverts are another necessary feature of roads. For decades, most western states required that culverts be designed to pass only the 25-year flood – a design that is statistically predicted to have a 50% probability of overtopping within 17 years of installation. Currently, the minimum design requirement of most state forest practices rules is 50 to 100 years. While this improves passage for fish, it likely reduces the number of wetlands created incidentally by logging roads, and in some cases culvert replacements may incidentally shrink or entirely dry out some wetlands created long ago by logging operations on the uphill side of roads.

An increase in the magnitude, frequency, or duration of flooding in wetlands can have several effects, depending on initial conditions and various interacting factors (Adamus and Brandt 1990, Adamus et al. 2001):

- kills trees, leading to vegetation changes and/or an increased area of open water, as also happens when beaver dam a stream;
- drowns some herbaceous plants, particularly invasives and facultative indicators of wetland conditions, because many of those are associated with drier upland environments and wetland margins (Drinkard et al. 2011);
- either increases or decreases the exposure of plants to competitors and herbivores;
- increases the habitat space and habitat suitability for most waterbirds, aquatic invertebrates, fish, and some amphibians;
- usually increases the within-wetland richness of aquatic invertebrates and aquatic plants, thus increasing food web complexity (e.g., Wisinger et al. 1999, Duffy 1999, Ludwa and Richter 2000);
- connects wetlands, thus:
  - improving water circulation and growth rates of several plant species;
  - facilitating fish access to additional nursery habitat;
  - increasing the exposure of invertebrates and amphibian larvae to predators;
  - allowing mixing of gene pools of flora and fauna;

  - facilitating the spread of invasive plants.
- decreases oxygen and (usually) pH in wetland sediment and water;
- increases the emission of methane, a potent greenhouse gas;
- increases the bioavailability of phosphorus previously bound in soil organic matter (by enhancing decomposition) or in sediments (by decreasing the oxygen in sediments); this benefits both desirable and undesirable (nuisance algae) species;
- increases (by enhancing decomposition) the availability of nitrate for plant growth, but decreases nitrate by enhancing microbial denitrification processes;
- increases the risk of iron and manganese toxicity to plants, depending on soil chemistry (Barrick and Noble 1993).

Topographic variation on the order of a few centimeters can shape the composition and richness of the plant community by influencing the duration (Dicke and Toliver 1990, Merendino and Smith 1991, David 1996, Vivian-Smith 1997, Silverton et al. 1999), timing (Merendino et al. 1990, Squires and van der Valk 1992, Scott et al. 1996, 1997, Gladwin and Roelle 1998), and frequency of saturation (van der Valk 1994, Smith 1996, Pollock et al. 1998) in the root zones of wetland plants. In floodplain wetlands, tree seedling survival may be lowest during years when flooding occurs at or shortly after the beginning of the growing season, or where surface water in wetlands persists for more than 40% of the growing season (Toner and Keddy 1997).

The amplitude and rate of water level **fluctuation** also influences plant species composition, biomass, and germination in wetlands (Hull *et al.* 1989, Hudon 1997, Shay *et al.* 1999). Water table fluctuations induced in wetlands by creation of impervious pavement, although far more profound and permanent than those possibly caused by logging, have been implicated in non-forest studies for facilitating the spread of invasive plants such as reed canary-grass (*Phalaris arundinacea*). Among 26 forested wetlands studied in the Seattle area, wetlands with mostly emergent or shrub vegetation whose water levels fluctuated more often and/or with greater amplitude had fewer plant species. In those wetlands, greater water level fluctuation decreased plant richness most noticeably during the early spring (Cooke and Azous 2000).

Plants themselves (especially particular configurations of woody and other deep-rooted plants, Loranty et al. 2008) can amplify runoff-related water level fluctuations in wetlands by transpiration processes, at a scale of hours, days, and seasons (Dubé et al. 1995, Jutras et al. 2006). In wetlands subject to periodic anoxia, increased water level fluctuations also can facilitate the denitrification process, resulting in loss of nitrate as nitrogen gas.

For streams, a few studies from the PNW have examined biological correlations with timber harvests. The most comprehensive analysis, conducted by Herlihy et al. (2005), analyzed invertebrate data from 167 randomly-selected headwater sites in Oregon with different degrees of timber harvest. At that scale, they found no statistically strong relationships between invertebrate index of biotic integrity (IBI) and logging activity (past or present), presence/absence of fish, catchment size, or ecoregion. All four severely impaired sites and five of the seven sites with moderate impairment were lower altitude, low gradient streams with evidence of agricultural activity in their catchment or riparian zone. See section 3.3.3 for additional information on biological effects of timber harvesting.

3.1.5 Summary and Data Gaps: Water Regime Effects

In this review, no studies were identified that measured directly the effects of various forest practices in the PNW on water regimes of wetlands. Specific data gaps are described by the hypotheses in part 2 of Appendix A.

Tentative inference may be made from the dozens of studies that have investigated logging effects on hydrology of streams in the region (summarized by Winkler et al. 2010) and studies not from this region yet focused on timber harvests in or near northern wetlands (Palik et al. 2001, Palik and Kastendick 2010, Hanson et al. 2009, 2010, Kolka et al. 2011). Collectively, these suggest that timber harvests in most PNW locations result in a rise in local water tables and greater water yield. The degree to which these effects extend off-site and influence wetland functions depends on local geomorphic conditions and remains unquantified. Construction of forest roads through wetlands displaces water that otherwise is often stored in wetlands. Forest roads also can hinder the downslope passage of surface water, especially during major storms. This potentially alters the timing and reduces the amount of water reaching wetlands immediately downslope, while potentially creating or expanding wetlands that adjoin the road on its uphill side.

3.2 Effects on Water Quality, Water Quality Function, and Soils

Wetlands have water quality **conditions**, and they have water quality **functions**. This is an important distinction. The concentration of nutrients is a condition, whereas the rate and capacity of a wetland to retain nutrients is a function. Forest practices can affect water quality condition with only minimal effect on water quality functions, and conceivably, can affect water quality functions with effects on water quality condition that are immeasurable over normal time scales. Both the condition and the function that are termed "water quality" are strongly influenced by soils. For example, when some soil types receive runoff, they release dissolved carbon into the surface water, affecting water quality *condition*. If that soil carbon is not replenished, or if soils are heavily compacted by machinery, the capacity of the soils to *remove nitrate (a function*) may decline over time. Forest practices can affect both the concentration of dissolved carbon in surface water, *and* the capacity of soils to influence that. A few studies have examined water quality *conditions* of harvested wetlands or wetlands with tree harvest nearby, but none have directly measured harvest-associated changes in the usual capacity of wetlands to remove nitrate, adsorb phosphorus, detoxify pesticides, or provide other water quality *functions*.

Popular media often describe wetlands as "kidneys on the landscape" for their capacity to remove pollutants. For example, one headwater forested wetland in British Columbia removed 27% of the organic nitrogen it received annually (Hill 1991). In Ontario, a 0.2-ha forested wetland that covered only 4% of its catchment removed 65-100% of surface water nitrate inputs (Spoelstra et al. 2010). However, like actual kidneys, wetlands have their limits. Excessive loading with some types of pollutants not only degrades water quality condition in a wetland, it can also diminish the future ability of the wetland to maintain or improve water quality. For example, many depressional wetlands are excellent for retaining whatever suspended sediment reaches them in runoff, but deposition of too much sediment will, over time, fill in a part of a depression, causing it to retain less runoff for shorter periods, and thus limit the wetland's future capacity to remove sediment (Whigham and Jordan 2003, Leibowitz 2003). Potential cumulative effects on a wetland's functional capacity to process pollutants were discussed by Hemond and Benoit (1988).

Vegetated buffers are widely used to maintain water quality of streams and wetlands. A common perception is that the main reason buffers are effective is because their vegetation takes up excessive nutrients in runoff from higher areas. However, plants in buffers usually retain for only a few months the nutrients they take up, returning most nutrients to the soil or exporting nutrients to the water column (and thence downstream if the wetland has an outlet) as the plants decay at the end of the growing season.

The most important reason buffers are effective is simply that they -- unlike clearcuts and other disturbances -- are usually not significant *sources* of additional sediment and nutrients. If they were a source, this would be the worst location to be a source, in terms of risk to other water bodies. That is because the low areas closest to streams and wetlands are the places where water tables tend to first intercept the ground surface during and after a storm (Walter et al. 2000, Buttle et al. 2004, Creed et al. 2008 a, b; Qiu 2009 a, b), thus connecting ephemeral drainageways and accelerating the downhill transport of sediment and nutrients that might originate from harvest. As one hydrologist (Walter et al. 2009) puts it:

> "A common conceptual assumption is that buffers "intercept" and treat upland runoff. As a shift in paradigm, it is proposed instead that riparian buffers should be recognized as the parts of the landscape that most frequently **generate** storm runoff. Thus, water quality can be protected from contaminated storm runoff by disassociating riparian buffers from potentially polluting activities."

Nonetheless, vegetation plays an important role in buffer effectiveness. Roots help maintain the capacity of soils to allow water to infiltrate, plants transpire sufficient soil moisture to keep soils unsaturated and receptive of runoff, carbon added by plants supports microbial communities essential for removing some pollutants, and shade cools runoff or a wetland's water directly.

Although discussions of buffer design typically focus primarily on the buffer's width (mainly because it is quickest to measure in a standard manner), several other buffer characteristics often have a greater effect on a buffer's effectiveness (Mayer et al. 2005, 2007). These include ground surface roughness, water source, flow pattern (concentrated vs. diffuse), slope, soil texture and organic content, location of the buffer relative to major paths by which water enters the wetland, contributing area size relative to buffer size, amount and dosing rate of the pollutant, vegetation type and density, and aspect (longer growing season on south-facing slopes promotes runoff retention and pollutant processing, provided soils are not dried out to less than 70% saturation by those warmer conditions; Kim et al. 2007, Hefting et al. 2006).

Of the factors named above, flow pattern is perhaps the most important. It is not unusual to find much of the runoff from harvest operations following gullies, ditches, and other semi-confined flow paths with sparse ground cover despite those gullies and ditches being located under a well-formed tree canopy constituting the required buffer. Under such conditions, the buffer is often an ineffective filter and detains the downhill movement of sediment only until the next storm (Dosskey et al. 2002, 2010). Thus, vegetated buffers are most effective in protecting the quality of wetlands when major inflows are diffuse (surface sheet flow or subsurface lateral flow) rather than concentrated in rills and gullies (Dillaha et al. 1989, Dosskey et al. 2001, Wigington et al. 2003). This depends on typical rainfall patterns (steady drizzle vs. concentrated in storm events, Lee et al. 2003) as well as soil type (coarser soils tend to promote infiltration and less gullying), extent of drainage alterations such as ditches that concentrate runoff, and slope (Abu-Zreig 2001, Mancilla et al. 2005). Within a buffer, spatial variation is often high in regard to the capacity of the soils and vegetation to remove nitrate (Hefting et al. 2006). In one study, only 9-18% of the vegetation in a buffer was actually in contact with runoff, due to the buffer's topography. Although under uniform flow the buffer could potentially remove 41-99% of sediment, the actual removal rate was 15-43% (Dosskey et al. 2001). Buffers in rural New York were found to be ineffective when crossed by small roadside ditches that were not buffered but were connected to pollution sources (Madden et al. 2007). If the sole purpose of a wetland buffer is to protect a wetland's water quality, then prescribed buffer widths might be reduced where the buffer slopes *away from* the wetland, or where uplands otherwise fall outside the path between the harvest area and the wetland. Such non-contributing areas do little or nothing to intercept polluted runoff that otherwise would reach the wetland.

Small buffers that are expected to bear responsibility for processing runoff from very large contributing areas tend to be ineffective, because storm runoff quickly overwhelms their processing capacity (Misra et al. 1996, Creed et al. 2008, Tomer et al. 2009). Not all buffer studies have found the ratio of buffer area to contributing area to be a good predictor of buffer effectiveness, but authors of those that have suggest the vegetated buffer acreage should be at least 15% of the acreage of its contributing area, especially the part of the contributing area that is capable of generating polluted runoff (Leeds et al. 1994).

For pollutant removal or maintaining water temperature, no data exist that indicate buffers dominated by non-native plant species are less or more effective than ones dominated by native plants. However, one relationship that does appear to be relatively well-documented is that wooded buffers dominated by nitrogen-fixing shrubs such as red alder (*Alnus rubra*) tend to be sources, not sinks, for nitrate (a potential pollutant) during at least some seasons of the year, and thus may be ineffective as buffers where the primary intent is to protect a particular wetland from overenrichment.

3.2.1 Forest Practices Effects on Wetland Temperature and Dissolved Oxygen

3.2.1.1 Tree Removal Effects

Most regional literature on impacts to stream temperature from timber harvest was summarized by Moore et al. (2005), Pike et al. (2010), and Tschaplinski and Pike (2010). In streams flowing through 40 small forested watersheds in the Olympic Peninsula, mean daily maximum temperatures averaged 58.1°F (14.5 °C) and 53.8°F (12.1°C) in logged and unlogged watersheds, respectively, even 40 years after harvest (Pollock et al. 2009, 2010). Diurnal fluctuations also were greater in the harvested watersheds, averaging 3.0 °F (1.7 °C) compared to 1.6 °F (0.9 °C) in the unharvested. Another study on the Olympic Peninsula (Murray et al. 2000) focused on just two watersheds with partial harvests (7-30%). Compared with an unharvested watershed, summertime maximum was elevated by 6.3° F (3.5° C) and became more variable. By 11-15 years after harvesting, stream temperatures were almost back to pre-harvest levels.

The proportional amount and pattern of shade in a watershed in some instances has an equal or greater cumulative influence on wetland and stream temperature than does shade from vegetation closest to the water (Brosofske et al. 1997, Sridhar et al. 2004, Stephenson and Morin 2008). For example, a study in Oregon (Beschta and Taylor 1988) found highly significant ($p < 0.01$) relationship between a cumulative index of forest harvesting and maximum stream temperatures. In a study on the Olympic Peninsula of Washington (Pollock et al. 2009, 2010), average daily maximum temperature depended on the amount of clearing in both the watershed as a whole and in just the parts of the watershed near the streams. The amount of recently clear-cut riparian forest (<20 year) within ~2000 ft upstream ranged from 0% to 100% and was not correlated to increased stream temperatures. The probability of a stream exceeding the temperature standard increased with increasing amount of the watershed harvested. All unharvested sites and five of six sites that had 25-50% harvest met the temperature standard. In contrast, only half the sites with 50-75% harvest and 2 of 9 sites with >75% harvest met the standard. Many streams with extensive canopy closure still had higher temperatures and greater diurnal fluctuations than the unharvested basins, indicating that that the impact of past forest harvest activities on stream temperatures cannot be entirely mitigated through the establishment of riparian buffers. Additional information on vegetated buffer effects on stream temperature is provided in section 3.2.1.3.

In British Columbia, a paired watershed study (Feller 1981) examined effects of clearcutting and slash burning on stream temperature and found maximum annual stream temperatures were 5.4 - 9.0 °F (3–5 °C) higher in the harvested watershed. The increase persisted 7 years in the unburned portion of the clearcut watershed and somewhat longer in the area where clearcutting was followed by slash burning. Clearcutting increased winter water temperatures, whereas slash burning caused a decrease in winter temperatures. Several other paired-watershed studies in British Columbia and Washington (Dong et al. 1998, MacDonald et al. 2003, Moore et al. 2005) have found that maximum daily water temperatures increase up to 10.8 °F (6°C) after clearcutting, and require at least 10 years to return to pre-harvest levels.

The severity of clearcutting's impact on water temperatures may depend partly on the amount, type, and configuration of shrub and ground cover that remains in the cutover area immediately after tree harvest (Gravelle and Link 2007). But more importantly, especially in most headwater areas and during low flow conditions, it will depend on the rates that groundwater and hyporheic flow are discharged into the stream or wetland (Story et al. 2003, Douglas 2006, Rayne et al. 2008). Ground water flowing into streams reduces stream heating by increasing the total discharge (measurably only in headwater areas) as well as cooling by conduction (Moore et al. 2005). Rough estimates of subsurface (hyporheic) flow, which often has a large groundwater component,can be made from observations of stream geomorphology in some settings (Kasahara and Wondzell 2003). In regions that experience winter ice cover, the simple presence of discharging groundwater is sometimes hinted at by the occurrence of unfrozen conditions later in the fall than is common among local waters of similar depth and circulation, and/or earlier thaw during late winter.

A study of two watersheds in the Oregon Cascades found that maximum stream temperatures increased 12.6 °F (7^{o} C) and occurred earlier in the summer after clear-cutting and burning in one watershed, but occurred after debris flows and patch-cutting in another. Hourly temperature fluctuations also became more pronounced in both watersheds. Stream temperatures in both gradually returned to preharvest levels after 15 years. Another Oregon study found that thinning a forest to a density of 80 trees/acre neither affected soil temperature in streamside areas nor the water temperature of the stream (Olson and Chan 2005). A recent BACI experiment in Oregon which compared streams with cut and uncut riparian buffers before and after timber harvest reported average summertime stream temperature increases of only 1.3 °F (0.7^{o} C) with an observed range of response from -1.6 to 4.5° F (-0.9 to 2.5^{o} C) (Groom et al. 2011).

Although it is apparent from the above that spatial variation of temperature has been studied widely in streams, little is known about the effect most wetlands have on stream temperatures – not even which types of wetlands are more likely to increase it or decrease it. As streams flow downslope, water temperature typically increases, especially if the slopes are south- or west-facing (Zwieniecki and Newton 1999, Gomi et al. 2006). Generally, mean annual water temperature in low-order streams also becomes warmer with decreasing shade from vegetation and topography, as well as with decreasing groundwater input (groundwater typically being cooler than surface runoff during much of the year), decreasing water depth, and increasing detention time (Moore et al. 2005). Wind exposure (turbulent mixing), substrate color (light absorption), and industrial effluent also can influence temperature (Moore et al. 2005). The influence of particular factors in this list varies greatly among streams (and presumably wetlands), so it is seldom possible to predict their net effect in a specific wetland. However, in one study in British Columbia where tree harvest raised water temperatures a maximum of 14.4 °F (8 ^{o}C) in streams, the temperatures in streams that originated in headwater wetlands increased a maximum of only 1.8-3.6 °F (1-2 ° C) (Rayne et al. 2008). Similarly, another study in British Columbia found that well-vegetated wetlands and lakes near the top of a watershed helped offset warming caused by harvest above them, thus allowing more rapid return to pre-harvest temperatures as the stream flowed downhill (Mellina et al. 2002). This can be expected in many places where wetlands are areas of focused groundwater discharge, because groundwater in summer is typically cooler than air temperature. Even in streams, groundwater influx was shown in one British Columbia study to be responsible for about 40% of the 5.4 °F (3 °C) cooling that occurred up to 200 m downstream from a timber harvest (Story et al. 2003).

In one of the rare studies on the effects of near-shore harvest on lake temperatures, Steedman et al. (1998) detected almost no change in 4 Ontario lakes ranging in size from 8 to 46 hectares. The lake with the largest warming -- up to 2.2 ° F (1.2 °C) -- had a south-facing shore, and the main effect was increased fluctuation in diurnal temperatures.

3.2.1.2 Logging Road Effects

Few if any studies have attempted to separate the thermal impacts of logging roads from those of timber harvest operations. Streams (and presumably wetlands) whose catchments have a greater extent of roads (road density) tend to have higher temperatures. A study of 104 streams in British Columbia found there is a 6-in-10 chance that the summer maximum weekly average water temperature will increase by 2.3 °F (1.3 °C) if road density in the contributing area exceeds 27 feet of road per acre and by 5.8 °F (3.2 °C) if road density exceeds 53 feet of road per acre (Nelitz et al. 2007).

3.2.1.3 Buffer Effectiveness for Maintaining Wetland Temperature

In coastal British Columbia, buffers of 10 m and 30 m appeared to protect headwater streams from significant changes in daily minimum, mean, and maximum temperatures. In streams with no buffers, clearcutting resulted in increases of 3.6-14.4 °F) (2-8 °C) (Gomi et al. 2006). After half the forest was removed in three small watersheds (<50 ha) in Quebec, a 20 m buffer appeared to protect stream temperature, with pre- versus post-harvest changes in summer daily maximum and minimum stream temperature of less than 1.8 °F (1 °C) and changes in diurnal variation of less than 0.9 °F (0.5 °C) (Tremblay et al. 2009).

From a series of sensitivity analyses using simulation modeling, Sridhar et al. (2004) reported that increasing the buffer width beyond 30 meters did not significantly decrease stream temperatures. Leaf area index and average tree height more strongly affected maximum stream temperatures. Another study found that maximum air temperature within a 30-m wooded buffer was only slightly cooler than in a 5-m wide wooded buffer (Meleason and Quinn 2004). In a British Columbia study of buffers along 13 headwater stream reaches, forests were clearcut and stream temperature associated with three riparian buffer treatments (30-m buffer, 10-m buffer and clear-cut to the stream edge) where compared with streams with uncut catchments (Kiffney et al. 2003). During some seasons, streams whose catchments had intact forest were cooler than streams bordered by a 30-m buffer. From a study of 20 small streams in western Washington, Dong et al. (1998) found that forested buffers of 16 to 72 m width did little to protect a stream from a 7.2 °F (4 °C) increase in air temperature that occurred in associated clearcuts during the middle of summer. However, they asserted that buffer width was not a significant variable in predicting air temperature above the streams, and that perhaps even the largest buffer was not sufficient to maintain air temperatures over a stream, because air temperatures over a stream with that buffer did not differ significantly from air temperatures over an otherwise analogous stream that lacked a buffer. In contrast, Groom et al. (2011) found that Oregon streams on private lands with minimal buffers experienced an average post-harvest increase in maximum temperature of only 1.3 °F (0.7 °C) with a range of -1.6° to 4.5 °F (-0.9 to 2.5 °C). Maximum, mean, minimum, and diel fluctuations in summer stream temperature increased with a reduction in shade, longer treatment reaches, and lower gradient. Despite the thinning of tree stands in 10- and 30-m buffers that adjoined timber harvests, stream temperatures were maintained for up to 200 m downstream in two stream reaches in British Columbia (Story et al. 2003).

The following (in italics) is quoting from a review by Liquori et al. (2008):

In California, Lewis et al. (2000) observed that cool tributary inflow (2.2° to 7.7° C below receiving stream) decreased the receiving water temperature for distances ranging from 3,000 to 35,000 ft (900 m to 10,700 m) downstream of the tributary junction. In Washington, Caldwell et al. (1991) found that headwater streams had minimal influence on the downstream water temperature because of the large size difference between headwater tributaries and receiving (typically fish-bearing) waters. Using a stream flow mixing equation and the relationship between distance from divide and discharge, they determined that a headwater stream could not affect the temperature in a typical fish-bearing stream by more than 0.49° C if the confluence of the receiving stream is more than 7 km (4.5 miles) distance from the watershed divide. Caldwell et al. (1991) reported that small streams are very responsive to localized conditions and that the longitudinal effect of any one headwater stream on downstream temperatures is limited to 150 meters or less. This study also evaluated the potential cumulative effects of multiple headwater streams feeding warm water into a fish stream. Based on a map analysis of tributary junctions, *they found that spacing between tributaries often exceeded 150 m and concluded that no cumulative effect was likely to occur.*

In summary, due to the varying and unpredictable effects of groundwater influx and other factors, no single, fixed-width buffer or canopy closure prescription will be adequate to maintain stream or wetland temperatures in all harvest operations (Richardson et al. 2012). Computer models (e.g., Oregon's Shade-o-lator, Boyd & Kasper 2003) are available for predicting water temperatures site-specifically, based on estimates of some or all of the key influencing variables, but are not routinely used.

3.2.1.4 Biological Effects of Altered Wetland Temperature and Dissolved Oxygen

Temperature is a fundamental regulator of biogeochemical cycles and biological productivity in wetlands. For example, as temperatures rise, less oxygen is capable of remaining dissolved in water. This critically affects the development, respiration, and metabolism of aquatic organisms, as well as a whole array of biogeochemical processes. Through this and other mechanisms, temperature (extremes, seasonal timing, and magnitude of daily fluctuations) can influence which species become established and persist at a particular location. Temperature can affect production of insects and algae; incubation of the eggs of invertebrates, fish, and amphibians; timing of fish rearing and migration; fish and amphibian susceptibility to disease; and many other factors (MacDonald et al. 2003).

The importance of water temperature is recognized by legal standardsadopted for streams. Cool waters (less than 68°F (°C), ideally less than 60°F (°C)) are particularly important to salmonid fish because at higher temperatures, less of the dissolved oxygen necessary for their survival (a minimum of 5 ppm is needed by most local fish) is able to remain in the water. In many PNW streams where vegetation cover has been reduced, water temperatures that are non-lethal but harmful to salmonid fish are common (Sullivan et al. 2000). Within a stream reach, the spatial extent and persistence (over weeks and months) of sublethal water temperatures may be more likely to influence salmon behavior and growth than the maximum temperature reached (Liquori et al. 2008).

Wetlands connected to rivers or in floodplains are used extensively by important temperature-sensitive fish such as coho salmon. Being highly mobile, fish can of course avoid channel segments with excessive temperatures unless trapped in pools by rapidly dropping water levels. However, when fish avoid areas due to such conditions, this reduces the extent of useable habitat and thus the number of fish that can exist in a stream overall. In Southeast Alaska, Thedinga et al. (1989) found that coho salmon in streams flowing through old-growth forest actually had fewer fry (age 0) than in clearcut and buffered streams, and the fry were smaller and emerged later. The fry remained longer in buffered streams than in clearcut or old-growth streams, but the size of coho parr (age 5 and older) did not differ among buffered, clearcut, or old-growth streams.

3.2.2 Forest Practices Effects on Sediment and Soils in Wetlands

Sediment inputs to wetlands from soil erosion are likely to increase as a result of ground disturbance associated with harvest (especially on steep slopes with erodible soils). Sediment inputs also increase as a result of increased windthrow of remaining trees, construction of logging roads and related facilities, and dust and runoff from those roads (Grayson et al. 1993, Kreutzweiser and Capell 2001). Processes that influence the transport of eroded sediment from all these sources are key to determining accumulation rates in wetlands (Western et al. 2001, Stieglitz et al. 2003). Hydrologic connectivity between topographically low areas such as wetlands is likely to increase during storms if logging-associated loss of tree transpiration and interception causes the water table to surface more often or for longer as described in section 3.1 (Wigington et al. 2005, Bracken and Croke 2007, Jones et al. 2009, McGuire and McDonnell 2010). Several models may have a capacity to predict the duration and extent of connectivity between wetlands and other surface waters (see review by Golden et al. 2014).

Being low points in the landscape, wetlands also tend to accumulate sediment. If a wetland neither has an outlet nor is on a floodplain, that sediment accumulates over time. In large enough amounts it can fill a wetland, ultimately converting it to upland. In other situations, excessive sediment can reduce the rate of exchange between ground and surface water (Moore and Wondzell 2005), reduce the water detention time, and harm some wetland plants and animals as described in section 3.2.2.4. On the other hand, when sediment is deposited along margins of deep open water areas (e.g., 1-2 m), it eventually provides a substrate for rooting of emergent plants, creating new wetlands or allowing existing fringe wetlands to expand outwardly. This pattern is prominent along lakeshores and sheltered estuaries.

3.2.2.1 Tree Removal Effects on Sediment in Wetlands

Studies of sediment and/or nutrient export from timber harvest operations were compiled and reviewed by Feller 2005, Gomi et al. 2005, NCASI Forest Watershed Task Group 2003, Gomi et al. 2005, Croke and Hairsine 2006, Liquori et al. 2008, Kreutweiser et al. 2008, Malik and Teichert 2009, Neary et al. 2009, and Pike et al. 2010. In western

Washington, watersheds containing less than about 15% forest, as a result of soil disturbance associated with logging or urbanization, have significantly more exceedences of water quality standards for suspended solids (Ludwa 1994). However, in many situations the loads of sediment generated specifically by timber harvests (not the associated roads which are discussed in section 3.2.2.2) appear to be relatively modest.

Erosion is typically greatest just after harvesting and road construction, and decreases soon thereafter (Megahan and Kidd 1972). For example, monitoring by MacDonald et al. (2003) determined that sediment levels in runoff returned to pre-harvest levels within two years of decommissioning of a logging road. Recovery may take longer where there are steep slopes (e.g., greater than about 30%) and catastrophic landslides, or where suspended sediment is composed mainly of very fine particles (e.g., clay, Duncan and Ward 1985), or where it is transported during storms in gullies, ditches, skid trails, and drainageways rather than as overland sheet flow. The propensity of some logged areas, during the post-harvest period of rising water table, to generate landslides that can either fill or create wetlands permanently has been well-documented (see review by Jordan et al. 2010). Landslide risk depends on slope, soil type, position in watershed, precipitation, residual cover, groundwater flow patterns, and other factors (Wemple et al. 2001, Jakob et al. 2006).

A survey of nearly 200 harvest units in Oregon and California, where logging had occurred 2-18 years previously, found only 19 occurrences of sediment-transporting gullies, rills, or sediment plumes. Only 6 of those occurrences (nearly all associated with skid trails) were connected to streams (Litschert and MacDonald 2009). Another survey of 300 harvest sites in California found only 37 incidences where erosion was evident in riparian areas that adjoined the harvests (Cafferata and Munn 2002).

A sediment dosing experiment involving two small streams determined that the streams stored a large proportion of the sediment washed from a road surface (Duncan et al. 1987). In no instance did either stream transport more than 45% of the added material to their mouths, distances of 95 and 125 m. Added sediment <0.063 mm in size was transported efficiently through the systems at all but the lowest flows tested. Material in the size ranges of 0.5 and 0.063 mm and 2.0- 0.5 mm in size was retained at progressively higher rates, and no more than 10% of the sediment in the coarser size category transited the entire stream length. In-stream woody material temporarily retained much of the sediment. A study of 6 ephemeral streams in Oregon found 60-80% of suspended sediment (size range= 1.6-53 μm) was removed from the water column over a 75 m stretch at moderate input levels (Dieterich and Anderson 1998). Even greater proportional retention would be expected in wetlands because of their usually flat topography. Also in Oregon, 9 years of monitoring the sediment output from an 8000-ha block of intensely managed forest indicated that no long-term changes existed in sediment yields, despite 180 km of roads being constructed and 3400 ha of old-growth forests being harvested from slopes averaging over 60% (Sullivan 1985). In one Washington watershed where turbidity in streams was high following logging and road construction in the 1970s and 1980s, turbidity had returned to pre-harvest levels by 2000 even with continued active forest management. The improvement was not attributable to interannual changes in flow (Reiter et al. 2009). It is possible that harvests might have elevated the stream turbidity and suspended sediment loads at more localized areas within the watershed and/or for short but ecologically significant periods in the first few post-harvest years.

Three types of soil disturbance can occur in association with timber harvests (Miller et al. 2001):

- Compaction: the process by which the soil grains are rearranged to decrease void space (particularly large pores) and bring them into closer contact with one another, thereby increasing the soil's bulk density;
- Puddling (also called liquidification): the destruction of soil structure usually by churning or kneading action of wheeled or tracked equipment;
- Displacement: the act of moving soil laterally from narrow ruts or wider areas.

The destruction of soil structure in puddled soils inevitably results in compaction. However, a soil can be compacted with only minor structural change. Even if soils are puddled without compaction, they will self-compact as they dry (Chancellor 1977). Soil compaction from harvesting equipment creates small depressions (rutting) and reduces the capacity of precipitation to infiltrate downward in soils. This sometimes creates small wetlands, but it also redirects runoff and accelerates the passage of water through the landscape, potentially increasing erosion, allowing less time for retention of suspended sediment, and resulting in sedimentation of downslope waters. Most wetland soils are sensitive to even minor compaction and other disturbance of their surface layer. Organic and clay soils are particularly susceptible to compaction, puddling, and rutting. In western Alberta, Corns (1988) estimated that compacted soils would require 10 to 21 years to return to pre-disturbance conditions, with surface layers requiring the longest.

Soil is least disturbed when low volumes of timber per unit area are removed per unit area. In Ontario, selective cutting of up to 50% of the timber in various harvest units appeared to result in no observable increase in sediment delivery to adjoining riparian areas (Kreutweiser and Capell 2001). Also, as noted in the review by Miller et al. (2004), soil is least disturbed when low when yarding systems are used that primarily lift rather than drag logs (e.g., skyline, balloon, or helicopter). Balloon or helicopter logging compacts or disturbs soil deeply in only 2-5% of the logged area (Dyrness 1972; Bockheim et al. 1975, Megahan 1988). In contrast, where cable systems do not provide full suspension or lift, the area of disturbed soil can range up to 45% of the logged area. Because full-suspension yarding methods are more costly, skidding logs with tractors, especially rubber-tired tractors, is more common where slopes are less than about 30% and skidding distances less than about 200 m (Miller et al. 2004). The amount and types of disturbance also vary depending on size and amount of logging residue, irregularity of terrain, moisture content at time of traffic, and inherent resistance of each soil to disturbance. Charts for assessing the susceptibility of a particular site to soil compaction were published by the B.C. Ministry of Forests (1999). Under some conditions when prescribed fire or slashburns are used, those practices decrease the capacity of soils to allow infiltration of precipitation, i.e., makes them more water-repellant (Certini 2005).

Soil compaction usually reduces plant cover and growth of surviving plants (Greacen and Sands 1980, Miller et al. 1996). Tree growth sometimes does not recover for several decades (Wert and Thomas 1981, McNabb et al. 2001). However, especially in coarse-textured soils, compaction sometimes increases growth of the surviving tree seedlings by improving near-surface moisture retention and reducing vegetative competition. Arthropods, fungi, nematodes, and/or microbial populations are generally lower in compacted than in reference soils (Smeltzer et al. 1986, Dick et al. 1988, Dexter 1978, Whalley et al. 1995), and overall biological activity is reduced (Dulohery et al. 1996). Resulting decreases in nutrient mineralization and availability may also contribute to reduced root growth (Phillipson and Coutts 1977). In addition, compaction-related changes in soil oxygen availability and root physiology can increase the incidence of soil pathogens (Jacobs and MacDonald 1990).

From a study of forest soils in the Rocky Mountains of Alberta under 3, 7, and 12 cycles (individual loaded trips) of skidding with mostly wide-tired skidders, McNabb et al. (2001) found that soil compaction occurred only when the soils were at or wetter than field capacity. A significant increase in bulk density due to skidder activity did not affect field capacity, permanent wilting point, and available water holding capacity. That was because the changes in soil porosity were essentially confined to the macro (larger) pore space while the micropore space remained unaffected (Startsev and McNabb 2001).

3.2.2.2 Logging Road Effects on Sediment in Wetlands

Forest roads, landings, and skid trails have been repeatedly implicated as the primary source of soil erosion and sediment runoff from silvicultural operations, e.g., Megahan and Kidd 1972, Schuldiner et al. 1979, Toews and Henderson 2001, Rivenbark and Jackson 2004, Jordan 2001, 2006.

Surveys of logging roads in western Washington and Oregon found that 42-66% of road drainage points discharged to hillslopes with no delivery to streams, 28-35% delivered directly to streams, and 17-28% delivered to streams via gullies (Bilby et al. 1989; Bowling and Lettenmaier 1997; La Marche and Lettenmaier 1998; Wemple et al. 1996). More importantly, however, these same or similar studies found that only 17-35% of the total road mileage contributes sediment to the stream system (Bowling and Lettenmaier 1997). In at least one logging operation where best management practices for sediment were aggressively implemented, the rates of sediment delivery from roads to streams were very low (0.05 to 0.2 tons/km^2/year) despite high road density. Thus, reducing the road mileage that discharges to streams and wetlands – not merely reducing total road mileage or density – can reduce the quantity of sediment that reaches streams and wetlands (Miller et al. 2001).

Road-related soil losses can be reduced with the use of several best management practices (Hynson et al. 1983, Welsch et al. 1995, Ice 2009, Ice and Schilling 2012). These include immediately vegetating cut and fill slopes (Swift 1988, Burroughs and King 1989), improving road surfaces (Reid and Dunne 1984), dispersing water onto hillslopes via broad based dips, turnouts and relief culverts (Luce and Black 1999), scheduling traffic during dry periods (Bilby et al. 1989), and providing vegetated buffers (Swift 1986).

3.2.2.3 Buffer Effectiveness for Protecting Wetlands from Excessive Sediment

Many studies have shown that sediment retention is greatest in the first 5-20 ft of a well-vegetated buffer, that is, the most uphill portion which is closest to eroding soil (Polyakov et al. 2005, White et al. 2007). A study in Georgia (White et al. 2008) found that under ideal conditions, even the finest sediment particles were removed within about 15 m (49 ft) as long as sheet flow, rather than gully transport, was the primary mechanism that transported the sediment. A review of 80 studies of buffers (Liu et al. 2008) concluded that for retaining sediment, vegetated buffers of about 10 m (33 ft) are usually optimal where sheet flow conditions predominate because additional widths retain only slightly more sediment. Similarly, a study of 22 logging sites in the Pacific Northwest reported that 10 m wide buffers were effective in eliminating chronic delivery of sediment to stream channels within 2 years after the harvest operation, except where buffers were crossed by cable yarding routes or skid trails (Rashin et al.

2006). A study in Ontario (Kreutzweiser and Capell 2001) reported that riparian buffers made very little difference in the amount of sediment reaching streams downslope from a timber harvest, perhaps because their effectiveness was partially compromised by roads and trails that intersected them. Even where experimental studies indicate that nearly all sediment is retained by buffer vegetation, retention is not necessarily long-term, as storms at decadal or longer intervals can remobilize sediments and move them into wetlands and streams (Gomi et al. 2005). Also, if buffers are too narrow, buffer trees are more susceptible to soil-disturbing windthrow (Pollock and Kennard 1998, Lewis et al. 2001). This can increase sediment delivery to streams and wetlands, especially as water tables rise in response to temporary elimination of transpiration losses from those trees.

3.2.2.4 Biological Effects of Sediment in Wetlands

Although no federal or state agencies have established numeric standards addressing sediment and nutrients in wetlands specifically, numerous studies have documented both their beneficial and adverse effects on species and functions in lakes and streams. In wetlands, some organisms might be expected to tolerate or even require higher concentrations of sediment and nutrients than is the case with stream organisms, but no studies have addressed that. What is known is this: wetlands in the Pacific Northwest whose watersheds have been partially cleared of vegetation do tend to experience both increased loads of sediments (suspended solids, turbidity) and nutrients (mainly phosphorus and nitrate), and these wetlands simultaneously experience shifts in their aquatic biological communities (see sections 3.2.2.4, 3.2.4.5, and 3.3). Those shifts can be judged beneficial or harmful depending on the species or processes affected and one's perspectives. And whether the increased sediment and nutrients are the primary cause of biological changes, as opposed to associated changes in runoff amount and timing or other factors, remains undetermined. A series of dosing experiments with wetland plants in Pennsylvania found that sedimentation generally lowered community biomass, diversity, and richness, while enrichment increased community biomass (Mahaney et al. 2004, 2005). As expected, depressional wetlands accumulated more sediment than riverine and slope wetlands (Wardrop and Brooks 1998). Plant species typical of floodplain wetlands were less sensitive to sediment additions than plant species typical of depressional wetlands (Mahaney et al. 2004). In another dosing experiment, sediment deposition caused an immediate drop in alder photosynthesis (Ewing 1999). Wetlands that are sufficiently deep and permanent to support submerged aquatic plants are perhaps most sensitive to inputs of suspended sediment, because most of those plants (e.g., sago pondweed in ponds, eelgrass in estuarine wetlands) require very clear water (Davis and Brinson 1980).

3.2.3 Forest Practices Effects on Nutrients, Carbon, and Silvicultural Chemicals in Wetlands

3.2.3.1 Tree Removal Effects on Nutrients and Carbon in Wetlands

In the short term, when trees in a wetland are cut and the wood is moved to distant locations, the wetland loses carbon that had been introduced by photosynthesis, as well as nutrients that had been taken up from the soil and incorporated into wood. In addition, logging waste (e.g., foliage and woody debris) releases nutrients and carbon as it decays. With the higher water tables resulting from logging (see section 3.1), there is increased risk of nutrients and carbon, as well as chemicals applied for silviculture, being transported off-site in runoff. In some cases, logging roads can expedite the export process.

Timber harvests also affect biogeochemical cycles by altering nutrient sinks and sources, increasing in soil temperature and sometimes humidity, altering soil structure, and flushing of nutrients and dissolved organic carbon from organic surface soils into wetlands and other surface waters (Carignan and Steedman 2000). As is the case with stream temperature, the amount of nutrients and carbon exported from timber harvest sites to wetlands depends partly on the amount, type, proximity, and configuration of vegetative cover that remains in the cutover area immediately after tree harvest, its proximity to wetlands, type of equipment used (degree of ground disturbance and compaction from skid trails, landings, etc.), time of year when logging occurs, soil erodibility, slope, restoration actions, and width of vegetated buffers. These and other factors are reviewed in the context of forest practices by Brown 1980, Binkley and Brown 1993, Brown and Binkley 1994, Neary and Hornbeck 1994, Dissmeyer 2000, Neary 2002, Feller 2005, Gomi et al. 2005, Gundersen et al. 2006, Stednick 2008, Kreutzweiser et al. 2008, VanMiegroet and Johnson 2009, Neary et al. 2009, and Pike et al. 2010.

Concerns have sometimes been raised about whether forests can retain their fertility after several harvest rotations. If the forest landscape surrounding wetlands loses nutrients over time, so may the wetlands that receive runoff and groundwater from it, with resulting reduction in their biological productivity. Much of the nutrient content of forest soils is contained in organic matter in the duff layer (including leaves, down wood) and other parts of the upper soil profile. Thus, loss of organic matter after harvesting or site preparation can have profound effects on soil physical, chemical, and biological properties and reduce soil productivity (Perry et al. 1989, Powers et al. 1990, Everett et al. 1994, Harvey et al. 1994, Jurgensen et al. 1997, Bulmer 1998, Page-Dumroese et al. 2001). This is especially concerning in drier parts of the region, because ectomycorrhizal root activity at such sites strongly influences site productivity and is closely tied to soil organic matter (Jurgensen et al. 1990). Some species, such as western hemlock, produce more seedlings and their survival rate is significantly higher when the highly organic duff layer of the soil is intact, particularly in drier areas. Studies of post-harvest changes in forest soil nutrients were compiled and reviewed by Kreutweiser et al. (2008) and Pike et al. (2010). Harvest effects on biogeochemical cycling specifically in northern wetlands were reviewed by Lockaby et al. (1999).

Phosphorus

Phosphorus can be lost from post-harvest soils by downward leaching of soluble phosphorus from decaying organic matter, as well as by increased transport off-site in connection with higher post-harvest water tables and erosion of phosphorus-bearing soil particles and decaying organic matter. In southern Alberta, release of phosphorus (P) from soil organic matter was found to be very fast during the first post-harvest year and declined thereafter (Prescott et al. 1993). However, of the relatively few studies that have compared soil phosphorus fluxes before and after logging, most have shown no statistically significant change in soil phosphorus (Kreutweiser et al. 2008, Malik and Teichert 2009).

Phosphorus export from upland harvested sites to wetlands increases in association with timber harvests partly because phosphorus readily adsorbs to fine sediment particles. Those particles are easily carried into low spots (streams and wetlands). Thus, like sediment, phosphorus export from harvested areas to wetlands is affected by landscape position, soil properties, and post-disturbance precipitation patterns (Putz et al. 2003, Chanasyk et al. 2003). Phosphorus and nitrate are common components of fertilizers applied to some managed forests (see section 3.2.3.3), and phosphorus is a major component of chemical retardants applied to wildfires. If some types of forest soils are not intentionally fertilized, it can take decades or centuries after trees are harvested to return soil phosphorus -- a key component of forest fertility -- to pre-harvest levels (Trettin et al. 1997). Soil phosphorus is restored very slowly because its main source is erosion of phosphorus-bearing rock.

In the Blue Mountains of eastern Oregon, dissolved phosphate in runoff increased after a forest was clearcut (Tiedemann et al. 1988). However, post-harvest phosphate concentrations in a coastal Vancouver Island lake were less than before logging occurred (Nordin et al. 2007). Following timber harvest and road building in one northern Idaho watershed, there was no significant change in runoff phosphorus concentrations (Gravelle et al. 2009). In Quebec, lowland (large watershed) lakes with more clearcutting in their watersheds had higher concentrations of total phosphorus than lowland lakes that had little or no recent timber harvests (Carignan et al. 2000). Among 11 watersheds in central Alberta where harvests had removed 0-35% of the timber, there was a 40% increase in phosphorus concentrations in the lakes whose watershed area had more than 15% of the trees removed (Prepas et al. 2001). Increases were most pronounced in lakes that were shallow (mean depth = 3.1 m, range = 0.7–14.4 m) and had (a) weak thermal stratification, (b) large drainage basins, and (c) shorter water residence times. Cyanobacteria (blue-green algae) also increased in response to the nutrient additions, primarily in shallow lakes and wetlands. Buffer widths (20, 100, and 200 m) did not appear to influence lake response to harvest. When timber harvest is accompanied by wildland fires or perhaps by extensive slash burning, phosphorus as well as sulfate concentrations in receiving waters tend to increase (e.g., Carignan et al. 2000, McEachern et al. 2000).

Nitrate

Nitrate can also be lost from post-harvest soils by downward leaching of soluble nitrate. It can also be diminished by conversion to nitrogen gas if post-harvest water tables rise and thus improve (usually) the conditions suitable for denitrification. However, this might be partly offset where nitrogen-fixing plants like alders quickly become established, allowing the nitrogen component of soil fertility to recover at least partially (Murray et al. 2000, Rothe et al. 2002, Stednick 2008). Timber removal at one location in British Columbia resulted in higher levels of soil nitrate even 7 years after harvest (Hope et al. 2003). However, most studies (compiled by Kreutweiser et al. 2008) have found no statistically significant difference in soil nitrate before versus in the first few years after harvest. In one case nitrate losses apparently related to harvest did not occur until many years post-harvest (Hazlett et al. 2007). In an Alberta study, no effects of timber harvest on the amount of nitrate in soil and in surface and ground water were detected, whereas topographic differences were a strong predictor, with wetlands having greater nitrate concentrations (Macrae et al. 2006).

Unlike phosphorus, significant amounts of the otherwise-increased nitrate can be removed from shallow groundwater by microbially-mediated conversion of nitrate to nitrogen gas, i.e., denitrification. In fact, forested wetlands (Phillips et al. 1993) and hillslope seeps such as those commonly found in new clearcuts have been documented to be hot-spots for denitrification (Hill 1991, O'Driscoll and DeWalle 2010). However, if water tables rise further as a result of logging-associated loss of tree transpiration, causing more of the water to "surface" during storms, then the transport into downslope waters of nitrate – as well as phosphorus, sediment, and other substances -- is likely to increase (Vitousek and Melillo 1979, Reynolds and Edwards 1995, Creed et al. 1996, Devito et al. 2000, Gumiero et al. 2011).

With regard to nitrate, clearcutting in the Blue Mountains of eastern Oregon was associated with only minimal increases in nitrate concentration, which peaked at 0.52 mg/L (Tiedemann et al. 1988). In the Olympic Peninsula of Washington, partial harvesting in two watersheds (7 and 33% of watershed) was associated with higher nitrate in runoff, which the researchers attributed mainly to a post-harvest increase in nitrogen-fixing alders (Murray et al. 2000). In the western Cascades of Oregon, nitrate yields in a clearcut were highest in the third year following harvest (2.9 kg/ ha), whereas the peak concentration (0.005-0.010 mg/L) occurred in the fourth year after harvesting (Adams and Stack 1989). Differences among four watersheds with different treatments (uncut, shelterwood with 50% area removed, patch cut with 30% removed, and clear-cut) were negligible after 9 years. Another study in the Oregon Cascades found that nitrate concentrations increased six-fold in a clearcut treatment for 7 years after harvesting. Concentrations frequently exceeded 0.10 mg/L during a high-flow period (Harr and Fredriksen 1988). An earlier Oregon study of timber harvesting and broadcast burning (Martin and Harr 1989) found that clearcutting and shelterwood harvesting in combination with broadcast burning was associated with a 30-fold increase in stream water nitrate levels. Surprisingly, this was less than half the nitrogen input they estimated from precipitation. In an Oregon Coast Range watershed, nitrate increased from 0.70 to 2.10 mg/L on an area that was both clearcut and broadcast-burned, but concentrations returned to pre-logging levels after 6 years. On a patch-cut treatment, there were no associated changes in nitrate (Brown et al. 1973). However, following a 50% partial cut northern Idaho, Gravelle et al. (2009) found the concentration of nitrate increased significantly downstream. Nitrate downstream from the clearcut increased to 0.35 mg/L, also a statistically significant change. Statistically significant but smaller increases were also observed further downstream. Forest road construction did not increase nitrate runoff significantly.

In British Columbia, nitrate concentrations nearly doubled one year after logging began but persisted for only 2 years at low flows and for 7 years at high flows after harvesting activities ended (Scrivener 1988). Nitrate and ammonia/ammonium concentrations increased in a small coastal lake on Vancouver Island after logging near its shores. Highest concentrations occurred 2–3 years after logging, and returned to pre-harvest levels within 5–8 years (Nordin et al. 2007). Another study in British Columbia (Feller 1989) indicated that slashburning doubled the stream water nitrate output compared to clearcutting alone. Nitrate export rates from all western forest lands, excluding areas dominated by alder, have been estimated at 78-79 kg/km^2/year (Wise and Johnson 2011).

The nutrient consequences of post-harvest domination of many clearcut sites by alders can be considerable, as red alder is perhaps the region's most prolific wetland nitrogenfixer. Where a semi-open canopy persists, red alders enrich the soil, resulting in a more productive and/or species-rich understory plant community with potential benefits to deer and perhaps other herbivores (Tappeiner et al. 1991, Hanley and Hoel 1996, Hanley et al. 2006). As compared with other riparian trees, alders may support more terrestrial insects that become prey for salmon (Allan et al. 2003). In Alaska, one study found that alders added 164 kg N per hectare per year to floodplain soils. Alders can account for up to 70% of the nitrogen that accumulates during vegetation establishment in river floodplains (Ruess et al. 2009). Some studies (e.g., Compton et al. 2003, Monohan 2004) have theorized or reported increased nitrate inputs to streams in watersheds with extensive nitrogen-fixing alders, with possible benefits to productivity of aquatic invertebrates (Hernandez et al. 2005, Medhurst et al. 2010) and perhaps salmon. However, Edmonds and Tuttle (2010) failed to find the expected higher nitrate concentrations in streams bordered by the most extensive alder cover.

Multi-year studies of logging effects are confounded by interannual variation in temperature and precipitation, which can be major drivers of local nutrient flux. Response of nutrients to timber harvesting can be very site-specific (Kreutweiser et al. 2009). Also, the effects of timber harvest are not always separated from the effects of new roads associated with the harvests, and different levels and configurations of tree harvesting are seldom compared. There appear to be no measurements of phosphate or nitrate loads or concentrations in or around this region's wetlands determined in the context of studies of forest practices.

Data from over 300 small forested streams throughout the United States suggest that organic forms of nitrogen prevail in hardwood (deciduous)-dominated watersheds whereas inorganic forms of nitrogen prevail in conifer-dominated watersheds (Binkley et al. 2004). A shift from a conifer to a deciduous forest often makes soils less acidic (Sabau et al. 2010) as does a shift from cedar to hemlock (Messier 1993, Prescott et al. 1996). Such changes could have strong implications for soil fertility and forest productivity over the long term (VanMiegroet and Johnson 2009). In 13 watersheds surrounding Puget Sound, deciduous riparian forests with moderate disturbance produced organic matter loads similar to those of conifer forests, but compared with conifer forests, they had a greater prevalence of leaf material and produced 54% higher nitrate loads and 40% higher phosphorus loads (Roberts and Bilby 2009). Conversion from coniferous to deciduous vegetation also produced subtle shifts in timing of inputs. Differences in forest architecture also can influence rates of nutrient subsidies (Cadenasso et al. 2004).

Carbon/ Organic Matter

Carbon is essential to nearly all life, and forest practices can influence its amount and cycling rate in wetlands. In wetlands, carbon-rich plant tissues (foliage, roots, woody debris) accumulate at a rate that depends on water residence time, turbulence, temperature, light, nutrients, acidity, the type of plant material, the abundance of decomposing organisms, and other factors. In coniferous forested wetlands or landscapes, the input and decomposition of organic matter is distributed more evenly throughout the year than in deciduous forested wetlands and watersheds, thus providing a more reliable source of carbon to food webs (Richardson et al. 2005b).

Carbon can be exported from wetlands in dissolved or particulate form (leached and infiltrated downward, or washed downstream if the wetland has an outlet) or as a gas – carbon dioxide and methane, both from respiring organisms. In addition, fire and floods can cause net losses of carbon from a wetland in the short term. Streams and lakes in watersheds with proportionately large areas of wetlands tend to have relatively high concentrations of dissolved organic matter (e.g., Creed et al. 2008). Some individual wetlands are net exporters of dissolved carbon (e.g., Dalva and Moore 1991, Walbridge and Lockaby 1994, Fitzgerald et al. 2003) whereas others are effective for sequestering carbon (e.g., Pelster et al. 2008). In wetlands of Southeast Alaska, levels of dissolved organic matter increase with shallower depth to water table and cooler soil temperature (D'Amore et al. 2010).

Apparently no published studies have measured changes in complete carbon budgets of wetlands in connection with timber harvesting in the Pacific Northwest. Those that have measured part of the budget -- i.e., dissolved carbon loads and concentrations in receiving waters before and after harvest -- have generally documented short-term two- to five-fold increases in dissolved carbon in downslope streams or lakes, especially when clearcuts further upslope were on peat soils. These include studies by Plamondon et al. 1982, Hinton et al. 1997; Startsev et al. 1998, Carignan et al. 2000, France et al. 2000, Lamontagne et al. 2000, Steedman 2000, Pinel-Alloul et al. 2002, O'Driscoll et al. 2006, and Bertolo and Magnan 2007. Multiple runs of a mass-balance model of carbon cycling suggested that stream organic matter might be temporarily reduced up to 80% as a result of removing riparian vegetation which provides the usual organic matter inputs to streams in southwestern British Columbia. In contrast, a stream temperature increase of 3 ^{o}C that resulted from shade removal might, through a tendency to increase decomposition of instream organic matter, reduce instream organic matter by only 20% (Karlsson et al. 2005).

From the moment trees are harvested, a potential exists for much of the residual non-marketable organic material (sawdust, limbs, foliage) to be washed or blown into wetlands. Especially in depressional wetlands, that material will add to a considerable amount of decomposing material already there which was produced by plants within the wetland. Many aquatic invertebrates in wetlands feed on plant detritus, and the specific characteristics of the carbon in that detritus -- which vary by its source (upland vs. wetland plant species) but not necessarily by forest age -- can influence its processing by those detritivores (Palik et al., 2001, 2006). If accumulation of carbon from both upland and wetland sources outpaces the capacity of organisms in the wetland to break it down, the wetland may eventually fill with organic matter and become more acidic (e.g., Shepard 1994, Brais et al. 1995, Trettin et al. 1997, Cui and Trettin 2005 a, b, Tremblay et al. 2009, McLaughlin et al. 2011). If hydrologic conditions are also suitable, native mosses (e.g., *Sphagnum* spp.) proliferate and amplify the acidifying effect, sometimes reducing plant species richness (Anderson et al. 2007). In bogs and many coniferous wetlands on peaty soils, the tendency of the post-harvest acidic conditions to restrict nutrient cycling limits the growth of most trees (e.g., Prescott et al. 1996, Gale et al. 1998). Well before that happens, the decaying material can severely deplete oxygen in water and sediments,. The oxygen depletion will stress some aquatic life, either directly or by supporting the emission of ammonia and methane from flooded decaying organic matter. Methane also is a potent contributor to global warming. Its emission rate from wetlands is difficult to predict, and increases or decreases depending on site characteristics and specific forest practices (Bubier et al. 1993, Roulet and Moore 1995, Dulohery et al. 1996, Lavoie et al. 2005, Li et al. 2004, Cui and Trettin 2005a, b). Dissolved carbon exports from logged watersheds also can be a vector for mercury, transporting this metal to downslope wetlands and lakes where high concentrations of methyl mercury can accumulate in animal tissues and become toxic to aquatic life (Garcia et al. 2007).

At the same time, increases in dissolved carbon inputs to wetlands, as a result of harvests further upslope, may have some ecological benefits. That is because some forms of carbon, besides supporting food chains and diverse biological communities, help protect aquatic life from harmful effects of ultraviolet radiation, and bind some toxic metals (Kerr et al. 2008). Soil organic matter also supports the conversion of soluble nitrate to harmless nitrogen gas by microbial communities, especially in groundwater seeps and wetlands within meandering stream networks (Pinay et al. 2003). In the Snohomish River Basin of western Washington, adult coho abundance and median adult coho densities were greater where peaty soils were more extensive, a finding that applied at both the watershed and stream reach scales (Pess et al. 2002). Peaty soils are typically associated with wetlands in this part of the PNW. Because of their high carbon content, such soils can be assumed to be particularly sensitive to even small alterations of soil structure. Those changes can result in significant carbon losses and reduced fertility (Li et al. 2004). Soil temperatures that speed decomposition of peat soils also can increase the export of carbon as carbon dioxide.

Literature reviews by Johnson (1992) and Johnson and Curtis (2001) found that most studies (of upland, not wetland soils) reported no statistically significant changes in soil carbon after timber harvest. Whatever losses were noted were usually small (seldom more than 15%). More recently, a survey of 19 sites representing a broad range of tree species, soil types, and climate identified no statistically significant losses of soil carbon from mineral soils after logging (Sanchez et al. 2006). However, a modeling analysis predicted a steady decline in soil carbon over a 300-year period for boreal forest tree species. The decline was exacerbated when shorter harvest rotation cycles were modeled (Seely et al. 2002). A study of Michigan forested wetlands on mostly organic soils found no difference between harvested and unharvested wetlands 11 years post-harvest in terms of carbon, nitrogen, calcium, magnesium, or potassium in three strata of above-ground vegetation, woody debris, roots, forest floor, and mineral soil (Trettin et al. 2011).

3.2.3.2 Logging Road Effects on Nutrients and Carbon in Wetlands

The effects of logging roads on nutrients and carbon have seldom been separated from the effects of tree removal. As noted earlier, roads potentially have the additional effect of accelerating the transport of nutrients, carbon, and silvicultural chemicals from a harvest site to wetlands.

3.2.3.3 Silvicultural Chemicals and Wetlands

Fertilizers

Timberlands are commonly fertilized to increase stand volume and reduce rotation length. In Oregon, approximately 95,000 acres have been treated annually (Chappell et al. 1992, Anderson 2001). Water quality concerns are mostly related to the potential for transfer into wetlands and other water bodies of phosphorus and nitrate, and the heavy metal contaminants sometimes associated with fertilizer. Studies on the nutrient effects on aquatic systems of the fertilizers used in forest management in the Pacific Northwest were reviewed by Bisson et al. 2008, Anderson 2002, Pike and Perrin 2005, and Pike et al. 2010. Overall, 1-27% of the nitrogen applied to forests runs off into streams (Anderson 2002). Based partly on results from simulation models, Wise and Johnson (2011) concluded that fertilizer applications under certain circumstances could temporarily be an influential source of nitrogen to streams in the Cascades and the Washington and Oregon Coast Ranges. Fertilization is not a reliable or long-term way to increase tree growth in all settings. This may be particularly true when attempting to reestablish trees in formerly forested wetlands where inundation and other factors can be more limiting. Even in uplands, controlling weeds that compete with tree seedlings can be a more effective long-term strategy for encourging growth of regenerating saplings (Rose and Ketchum 2002).

Pesticides

Insecticides are applied to forests for insect pest control and in wetlands are a particular concern to aquatic invertebrates and fish. **Herbicides** are generally of lesser direct concern to aquatic animals. They are applied, sometimes in conjunction with prescribed burns, to kill vegetation that threatens newly planted seedlings, i.e., "site preparation." Herbicides not only kill plants, but in doing so they directly or indirectly increase the concentration of nutrients in runoff (Likens et al. 1970, Sollins et al. 1981, Neary et al. 1986, Davis 1987, Feller 1989, Simpson et al. 1997). Nitrate increases can be minimized by increasing the intervals between herbicide applications sufficiently to allow revegetation to occur, assuming the new growth is not predominantly alder.

The transport of herbicides and their degradation products was reviewed by Norris et al. 1991, Dissmeyer 2000, Michael 2000, and Clark et al. 2009. Naturally, the potential for herbicide drift into wetlands is greater when they are applied aerially rather than manually. The most mobile herbicides are those that are water-soluble, do not degrade quickly, do not volatilize easily, and are not easily sorbed by soils. The fat-soluble–water-insoluble herbicides are generally more toxic than those that are water-soluble. Most herbicides tend to dissipate quickly or are detoxified quickly by microbial communities or photolytic processes, and thus do not bioaccumulate. However, pesticide formulations and tank mixes (including adjuvants and surfactants) are constantly being changed by manufacturers and information on some components is proprietary. This often makes it difficult to compare the toxicity of current brand name pesticides and their degradation products with ones of the same name formulated in past years, for which toxicological test results may be available. A review by Macneale et al. (2010) highlights the fact that relatively little is known about sublethal and indirect effects of forest pesticides on salmonids in the PNW. Also, relatively little is known about *indirect and cumulative* effects of pesticides, fertilizers, and shortened rotations on soil acidity, fertility, and availability of trace elements that influence the species composition of aquatic and terrestrial plant and animal communities. Limited evidence suggests such effects could be significant in at least some local instances (Flueck and Smith-Flueck 2006).

Fire Retardants and Dust-control Compounds

Fire *retardants* are part of a suite of tools and options for managing wildland fire. At any given location in the region, fire retardants are unlikely to be applied more than once in many decades, yet they are noted because of their potential toxicity to aquatic organisms, including those in wetlands. They typically are applied aerially, which increases the potential for wetlands being exposed because many wetlands are not visible from the air under conditions of poor visibility that accompany wildfire (smoke, night-time). Retardants are primarily phosphate- or sulfate- based compounds and often are contaminated with various levels of heavy metals. Toxic concentrations can occur, particularly when light levels are high and soils adjacent to wetlands or streams are coarse-textured and contain little organic matter (Little and Calfee 2002). Fire *suppressants* are foams comprised of proprietary mixtures of sodium and ammonium salts, alcohol, ether, and sulfates. *Dust suppressants and soil stabilizers* are applied seasonally to many unimproved forest roads, especially during construction near settled areas. Dust suppressants include various salts, asphalt emulsions, oils, synthetic polymers, mulches, and lignins. Many are by-products and their exact composition is unknown (Piechota et al. 2004).

3.2.3.4 Buffer Effectiveness for Protecting Wetlands from Excessive Nutrients and Silvicultural Chemicals

Phosphorus

Nutrients that are most often targeted by vegetated buffers are nitrate and phosphorus. Literature on phosphorus retention by vegetated buffers was reviewed by Hoffman et al. (2009). The ability of vegetated buffers to attenuate phosphorus before it reaches wetlands or streams depends largely on the degree to which runoff between the harvested area and receiving waters is concentrated in drainageways on the land surface, as well as the form of the phosphorus. Where phosphorus is mainly attached to sediment (as often it is), then buffer widths sufficient for sediment retention may be almost as effective as those specified for retaining phosphorus (White et al. 2007). For example, despite the limitations of the Zhang et al. (2010) meta-analysis of field studies, they calculated 97-100% retention of phosphorus by buffers of only 66 ft (20 m) width. However, where phosphorus is mostly in dissolved form (i.e., dissolved reactive phosphorus, soluble reactive phosphorus, orthophosphate), and where many surface drainageways intersect the vegetated buffer, then buffers may need to be very large or may not be effective at all (DeVito et al. 2000, Prepas et al. 2001).

The capacity of a wetland or its buffer to retain phosphorus has been shown to decline with continued exposure to nutrient runoff. As phosphorus accumulates in soils, its continued storage becomes more precarious. As it accumulates beyond some unknown equilibrium threshold, it is more easily desorbed and exported (Dillaha et al. 1988). This has led some investigators to propose harvesting of trees and other vegetation from wetlands and their buffers as a means of delaying the oversaturation of these sites with phosphorus (Lee et al. 2000). However, potential benefits from such harvests could easily be offset by increased transport of remaining phosphorus to wetlands as a result of higher post-harvest water tables.

Nitrate

In the most comprehensive literature review and meta-analysis to date, Mayer et al. (2007) synthesized results from over 60 peer-reviewed studies of **nitrate** removal by buffers adjoining a wide variety of land uses (agricultural, urban, silvicultural) with diverse nitrate loading rates in temperate climates. They found that widths of approximately 5 m, 50 m, and > 150 m are needed to achieve, respectively, 50%, 75%, and 90% removal efficiencies for nitrate. A similar meta-analysis of buffer widths was published by Zhang et al. (2010) but was based on only 8 studies of nitrate removal by buffers. Zhang et al. reported that 91-100% of nitrate loads can be removed by a buffer of only 20 m.

One study in Ontario found that where a confining layer was situated beneath very coarse soils, buffers with widths of up to 175 m were required to remove 90% of the incoming nitrate (Vidon and Hill 2006). With regard to runoff specifically from harvest, after a Nova Scotia forest was clear-cut, nitrate increased in only one of 6 streams where riparian buffers were 20 m (66 ft) wide or wider (Vaidya et al. 2008). Selective cutting within the buffers reduced their effectiveness for removing nitrate. Whether a buffer is forested or contains only intact herbaceous vegetation appears to have little influence on its capacity to remove nitrate (Sabater et al. 2003).

Buffer widths required to protect water bodies from nutrient-bearing logging runoff might be assumed to be narrower than those needed for protecting water bodies from agricultural or urban land uses, because under many or most conditions, even clearcuts would be expected to release smaller amounts of nutrients than from the other land uses. However, most studies of nitrate transport and removal have been conducted in lowland agricultural watersheds rather than headwater forested watersheds (Ranalli and Macalady 2010). In southwestern British Columbia, Feller (1989) found that the transport of nitrate to streams from clear-cutting generated about 10 kg N/ha, as compared with clear-cutting plus slash burning (~ 20 kg N/ha) and with herbicide application in young plantations (~ 40 kg N/ha). For comparison, on the average one horse on 5 acres generates about 20 kg N/ha of nitrate (in manure), while suburban lawns and croplands are commonly fertilized at much higher rates.

Pesticides

A sufficiently wide vegetated buffer, when maintained next to a pesticide application area, can help reduce transport of pesticides into wetlands and streams where their application is not generally allowed (Reichenberger et al. 2007). However, buffers are not a panacea. A review of studies published on herbicide retention by buffers concluded that the ratio of the acreage of the herbicide source area to that of the buffer did not generally influence buffer effectiveness in the area ratio range of 5:1 to 45:1 (Krutz et al. 2005). Under calm wind conditions, densely vegetated buffers with a width of 10 to 33 ft reduced most of the drift of aerially applied pesticides before surface waters were contaminated (Brown et al. 2004). At higher wind speeds, either a 20-m wide hedgerow or the same hedgerow plus a 10-m wide dense tree stand was found to be effective in one study. However, runoff can still carry these substances to nearby water bodies. A pesticide runoff study in Georgia demonstrated retention of some pesticides by a 125-ft buffer containing a mature hardwood riparian forest (Lowrance et al. 1997, Vellidis et al. 2002). Computer models developed by Rodgers and Dunn (1993) suggested that buffers of at least 330 ft (100 m) width would be needed to retain and detoxify one common herbicide (atrazine) during times of peak runoff. This finding was confirmed in wetland studies by Moore et al. (2001), although Gay et al. (2006) found 92-100% of atrazine could be retained within 105 ft if vegetation was dense and calm wind conditions prevailed at the time of application. Even larger buffers would be needed to retain chlorpyrifos or metolachlor, insecticides which are directly toxic to aquatic life (Moore et al. 2001, 2002). Zhang et al. (2010) calculated that 93% of pesticide loads (all compounds lumped together) are retained by buffers of 30 m. Their analysis suggested that buffers wider than that would not appreciably improve the removal efficiency. The applicability to the Pacific Northwest of their conclusions may be limited due to this region having steeper slopes and greater annual precipitation than most of the studies included in their meta-analysis. Moreover, even near-trace amounts of some pesticides can have harmful biological effects (Vandenberg et al. 2012).

3.2.3.5 Biological Effects of Excessive Nutrients and Silvicultural Chemicals

Nutrients

No state or province has adopted numeric water quality standards specifically for wetlands. Moreover, neither the federal governments nor the states/ provinces have established nutrient standards for any water body type. This is the case despite the widespread recognition that if nutrient loads are excessive, wetlands – like any water body – can experience large blooms of algae that temporarily diminish dissolved oxygen, shade out aquatic plants, and stifle aquatic productivity. Even when massive algal blooms are absent, slight shifts in the dominant algal species as a result of canopy removal (e.g., from diatoms to filamentous green taxa; Shortreed and Stockner 1983, Danehy et al. 2007) and shifts in the nutrient content and types of leaf litter (deciduous or coniferous; Cummins 2002) have the potential to influence invertebrate and fish populations (Anderson 2002). Timber harvests have been reported to have had such impacts on lakes (e.g., Planas et al. 2000) so it is perhaps even more likely that wetlands, with generally smaller ratios of catchment area to wetland surface area, could experience such impacts. However, no strong evidence exists that shows which changes in leaf litter type or forms of algae should be judged to be "good" or "bad" for favored consumers at various seasons and at different points in post-harvest time.

Also, many of this region’s rare and threatened native plants are ones that inhabit only the most nutrient-poor wetlands (e.g., bogs, fens, alpine meadows). In general, organisms in nutrient-poor aquatic habitats are most likely to show large responses (positive or negative) to nutrient additions (Marczak et al. 2007). No studies exist to evaluate their vulnerability to runoff from fertilizer applications. For nitrate, forest fertilizer applications must be considered as additional inputs on top of apparently increasing inputs of wet and dry deposition of atmospheric nitrate. Also, some studies have suggested that elevated levels of nutrients might contribute to increased vulnerability of wetlands to invasive species, which reduce plant richness and can ultimately homogenize a region’s flora (Lougheed et al. 2008). A study of 16 Michigan depressional wetlands surrounded by various extents of development within 1 km indicated that wetlands with greater nutrient concentrations had greater cover of duckweed and less cover of other floating-leaved and submerged aquatic plants (Lougheed et al. 2008). However, nutrient increases typically accompany other impacts and in many cases evidence of their separate impact is unclear. Compared to wetlands with submerged aquatic plants, duckweed-covered wetlands have less dissolved oxygen in their water, greater phosphorus mobility, and greater capacity to remove nitrate (Veraart et al. 2011).

Another perspective on nutrients holds that certain favored animals (such as salmon) grow slowly in forested wetlands and water bodies which have little nutrient input. This is a concern to some consumers. In fact, some lakes, reservoirs, and streams in British Columbia are fertilized in an attempt to improve local fisheries (Pike and Perrin 2005). Historically, many streams and connected wetlands were fertilized naturally by spawned-out fish that brought in nutrients from the ocean. With the decline of many salmon runs, recently some agencies have attempted to intentionally add nutrients to streams, mainly in the form of salmon carcasses. In nearly all instances, concentrations in streamwater have been elevated for fewer than 100 meters downstream, and lasting biological benefits are uncertain. A single PNW study evaluated the possible effects of applied fertilizer on stream invertebrates (Meehan et al. 1975) but results were somewhat uninformative for a variety of reasons (Anderson 2002).

Many wetlands naturally convert phosphorus and carbon from inorganic to organic forms. The degree to which forest practices might influence or interfere with this function in wetlands is unknown, as is the ultimate effect of altering these processes, in terms of wetland food chains, productivity, and biodiversity. Decomposition of organic matter (plant detritus) is usually faster in cutover areas due to warmer temperatures and in some cases, greater soil moisture immediately post-harvest (Trettin et al. 1996).

Pesticides

Literature on the ecological effects of applying pesticides (especially herbicides) in forestry operations was compiled and reviewed by Ponder 2002, Solomon and Thompson 2003, Krutz et al. 2005, Thompson et al. 2006, and Clark et al. 2009. Reviews by Solomon et al. (2008) and Hayes (2011) debated the dangers posed by atrazine, and a review by Govindarajulu (2008) focused on glyphosate, the most widely-used pesticide in the region and one for which sublethal effects on amphibians have been detected (Relyea et al. 2005, Cauble and Wagner 2005, Thompson et al. 2006, Relyea and Jones 2009). The observed toxicity appears tied to adjuvants used at that time rather than the parent compound (Govindarajulu 2008). Herbicide concentrations in surface waters have sometimes exceeded permissible levels, but little is known of the levels of specific herbicides, associated surfactants, and their by-products that result in no harm to all life stages of non-target plants and animals. Even when applications of some pesticides result in concentrations that meet federal water quality guidelines, they have been shown to harm amphibians native to the PNW (e.g., Westman et al. 2010, endosulfan, azinphosmethyl, and diazinon impacts to Pacific treefrog). Amphibians and rare native plants may be especially vulnerable because vegetated buffers are not routinely required for protecting the ephemeral drainageways and temporarily saturated wetlands that are habitat for many species in these groups. Pesticide concentrations that appear to be non-toxic when applied at one time of year may be toxic at others, and impacts of repeated dosing and interactions with other environmental stressors are seldom evaluated (Thompson 2004, Jones et al. 2010). Exposure levels and probabilities also are seldom measured *in situ*, and depend on species life history relative to usual times of pesticide application, subsequent precipitation, natural concentrating factors such as water thermal stratification (Jones et al. 2010), and extent of dilution by runoff or groundwater from untreated areas.

Most studies have not found herbicides to be directly toxic to larger vertebrates at the dosages typically applied. However, indirect effects can occur, especially if herbicides are applied annually at the same location (Anthony and Morrison 1985, McComb et al. 2008). Indirect pesticide-induced changes can include alterations of food chains, cover, and the normal chemical environment in waters of forested wetlands (Chen et al. 2004). Herbicides such as glyphosate have been shown to mobilize potentially toxic metals such as copper from forest and wetland soils that already have naturally high concentrations as a result of local geology (Barrett and McBride 2006). This can harm salmonids and other aquatic life. In Alberta, herbicide treatments reduced winter ungulate and summer elk forage but enhanced summer forage for moose about 14% (Strong and Gates 2006).

Fire Retardants and Other Silvicultural Chemicals

The very limited information on ecological effects of fire retardants and dust suppressants was reviewed by Adams and Simmons (1999) and Giménez et al. (2004). An instance of a chemical dust suppressant damaging stream invertebrates and fish was reported by Ettinger (1987) but few other studies on these compounds have been published. More recently, a study that included some temporary wetlands (Angeler et al. 2006 a, b) found that one retardant (Fire Trol 934) reduced invertebrate diversity and was possibly responsible for decline of submerged aquatics due to reduced light penetration following the retardant's application.

3.2.4 Summary and Data Gaps: Water Quality Effects

It should be apparent from the aforegoing review that few if any peer-reviewed studies have measured directly the effects of various forest practices in the PNW on water temperature, sediment, nutrients, or silvicultural chemicals in **wetlands**. Specific data gaps are described by the hypotheses in part 3 of Appendix A.

Tentative inferences can be made from the dozens of studies that have investigated harvest effects on the water quality of streams (as reviewed, for example, by Moore et al. 2005). However, it must be recognized that wetlands generally tend to be more acidic, warm, and anoxic than streams (althoughmany exceptions exist). Removal of a forest canopy increases the temperature of surface water over variable distances downslope, including water temperature in wetlands. Increased water yield associated with tree removal, especially when accompanied by ground disturbance, increases the downslope transport of soil and its associated elements and compounds over variable distances (as reviewed, for example, by Liquori et al. 2008, Kreutweiser et al. 2008, Pike et al. 2010). Forest roads are potentially a source of suspended sediment to downslope wetlands, and this depends on their design and location. Under some situations, retention of vegetated buffers around wetlands can reduce the transport of sediment and chemicals into wetlands located downslope of forestry operations. As noted earlier, in one study from Michigan, no statistically significant differences were found in nutrients between harvested and unharvested wetlands 11 years post-harvest (Trettin et al. 2011).

3.3 Effects of Forest Practices on Microclimate, Vegetation, and Animals in Wetlands

3.3.1 Effects on Microclimate

Microclimate refers to conditions of temperature, moisture, wind, and sunlight measured at scales ranging from a few centimeters up to perhaps a kilometer. Microclimate is usually described for aboveground environments, but the concept can also be applied to water and soils. At the scales at which microclimate is measured, its components are influenced chiefly by:

- vegetation (including plant litter and down wood)
- topographic relief
- snow cover extent and duration
- groundwater exchange
- heat absorption/retention capacity of the substrate
- proximity to large water bodies or extensively bare areas

Microclimate contrasts with *climate* which is influenced mainly by elevation, latitude, marine proximity, and other factors measured mainly at continental or regional scales. Even in wetlands and forests unaltered by humans, microclimate varies spatially to a large degree as a result of natural disturbances such as windthrow (blowdowns), fire, and plant succession. Timber harvesting and replanting, logging roads, and herbicide applications can all affect wetland microclimate directly or indirectly, positively or adversely. These effects are described in this section with the exception of effects on water temperature, which are described in section 3.2.1. Studies of "edge effects" on microclimate within forests or above streams have been compiled and reviewed by Trombulak et al. 2000, O'Connell et al. 2000, Gucinski et al. 2001, Coffin 2007, Fahrig and Rytwinski 2009, and Daigle 2010.

Although microclimate is a major determinant of the species composition and local distribution of wetland plants and animals, relatively little is known about the microclimate requirements or preferences of individual wetland species. It appears that no single set of microclimate conditions is preferred by all wetland species (Keller et al. 1993).

Different forest practices as well as different natural disturbances can affect wetland microclimate to varying degrees, benefitting some species and harming others. Microclimate can also be altered by clearings created for buildings or roads, by natural phenomena (windthrow, landslides, streams), and by overgrazing of shrub and understory vegetation by deer and domestic animals. Microclimate conditions (humidity, temperature, etc.) and their natural variation have not been quantified among a wide range of wetlands within the PNW region.

Lakes, streams, and marine waters have been shown to influence the microclimate of adjoining upland areas at varied distances and consequently influence the upland flora and fauna (Olson et al. 2007). This effect has been estimated to extend only about 10 (Danehy and Kirpes 2000) to 20 m (Rykken et al. 2007) from the edges of headwater streams (channels less than ~1 m wide)and perhaps as much as 100 m from mid-sized streams (channels ~ 2 to 15 m wide) (Chan-MacLeod 1996). Effects on upland flora of increased air moisture resulting from overwater evaporation in a stream or wetland may be difficult to balance against drying effects of increased sunlight from a stream- or wetland-caused break in the forest canopy.

The capacity of wetlands to influence upland microclimate has not been documented in the Pacific Northwest. It likely depends on wetland size and configuration, vegetation extent and structure, water depth, groundwater influence, surrounding topography, season, and other factors. All of these factors can be influenced by forest practices.

When the area of a forest is reduced by harvests that remove all or most of the tree canopy, the forest's remaining vegetation is exposed to more sunlight and wind. This increases the evaporative water losses, reduces soil moisture, and potentially shortens the duration of inundation in some wetlands. However, as explained in section 3.1, in most cases the losses are offset by increased water availability resulting from timber harvest and associated reduction in water losses from transpiration.

Many estimates have been made of the distances over which timber harvests affect microclimate. Some have found that the influence of cleared areas on the microclimate of an adjoining forest extends about 50 m (~160 ft) into the forest but in extreme cases can extend as far as 500 ft (Dignan and Bren 2003, Ries et al. 2004, Moore et al. 2005, Hennenberg et al. 2008). Studies in western Washington found the light regime on the forest floor was affected ~100-200 ft from the edge, while humidity and air circulation were affected as far as ~800 ft into the forest (Chen et al. 1990, 1995).

A second study of riparian areas in western Washington suggested a wooded buffer of 31-62 m on both sides of a stream, comprised of about 75% canopy cover, might be necessary to approximate the natural microclimate gradients around the stream, and for some microclimate parameters, widths of up to 300 m might be necessary to maintain conditions (Brosofske et al. 1997). Edge effects (effects on microclimate of adjoining clearing) are most pronounced along south- and west-facing edges, perhaps suggesting the need for wider vegetated buffers there (Gehlhausen et al. 2000).

A third study from the region reported that most changes in light and temperature occurred within 20 m of an edge, and soil temperature reached normal levels 30 m from edges (Heithecker and Halpern 2007). The distance from edge within which the microclimate of a forested buffer was altered depended on forest structure and aspect, especially those conditions within 20 m of the edge of dense stands on steep terrain. Edge effects extended much farther (56 m) into forests on south- or west-facing slopes than into forests on north- or east-facing slopes.

Based on the limited amount of research published prior to 1993, the Northwest Forest Plan (FEMAT 1993) suggested that a forested riparian buffer having a width of about 0.6 times the potential height of tree species growing along a given stream might be adequate to maintain enough shade in that stream, at least for many small streams. Assuming a potential tree height of 200 ft, this translates to a shade buffer of 121 feet on each side of a stream. If maintaining shade at naturally-occurring levels in wetlands is considered important, the FEMAT guideline may be considered applicable to many wetlands.

A modeling simulation suggested that air temperature in a forest might sometimes be affected up to 230 ft from an edge; a warming of only 7 ^{o}F could change relative humidity exponentially from 94% to about 77% (Dong et al. 1998) with consequent effects on mosses, lichens, amphibians, and other organisms, many of which require sufficient humidity during certain phases of their development. Lichens and mosses have been affected by edge-induced microclimate changes extending at least 15 m into forested areas (Hylander et al. 2002, Stewart and Mallik 2006, Boudreault et al. 2008) and as far as 50 m (~170 ft) from the forest edge (Baldwin and Bradfield 2005, 2007). On Vancouver Island, even small (< 1 ha) 120-year-old conifer stands, surrounded by old-growth and containing much structure in the form of remnant large trees, did not contain the same abundance or composition of epiphytic lichens as did old-growth stands (Price and Hochachka 2001), although the difference cannot be attributed solely to difference in microclimate. The orientation of the edge that is created by forest roads or clearcuts can influence the impact on lichens and other plants (Johansson 2008) or not

(Heithecker and Halpern 2007). Effects on some lichens of removing the forest canopy can be reduced by making the forest edge a spatially "soft" transition that shifts gradually from dense forest to shrubs to short open vegetation (Stevenson and Coxson 2008).

Gradual loss of buffers as a result of windthrow not only alters the microclimate within wetlands, but also can alter their nutrient regimes and habitat suitability. Following the loss of lakeside buffers in Ontario, both as a result of harvest and windthrow, the average overwater wind speeds tripled and the thermoclines of those lakes deepened, although other factors influencing those could not be ruled out (France 1997). This potentially diminished the habitat available to coldwater fish and might have resulted in reduced phosphorus mobility in these lakes, and thus lower productivity.

A less direct way of considering microclimate is to examine the minimum size and configuration of a forested tract, patch, or leave-tree cluster (residual forest aggregate, green tree patch) required to adequately maintain microclimate of its interior (“core”). Heithecker and Halpern (2007) found that forest aggregates (groups of trees left standing after surrounding trees were cut) of only 1-ha, with normal tree densities and suitable configurations, were sufficiently large to contain areas which provided conditions of light, temperature, and soil moisture comparable to those in undisturbed forest, at least at a point in time that was 6-7 years post-harvest. That study also noted that the effects on microclimate of aspect (north- or south-facing etc.) were at least as strong as the effects of forest structure (e.g., percent retention).

Although most studies have examined *clearcut-associated* changes to the microclimate of adjoining forests, a few have investigated the effects of forest *thinning*, and have determined that thinning does alter localized air temperature as compared with unthinned stands, at least in the southern parts of the PNW region. In one case, thinning resulted in mean air temperature maxima above adjoining streams that were 1.8 to 7.2 ^{o}F (1 to 4 ^{o}C) warmer than where adjoined by unthinned stands (Anderson et al. 2007). However, where unthinned forested buffers of at least 15 m width were present, the daily maximum air temperature above an adjoining stream was virtually unchanged, and daily minimum relative humidity was less than 5% lower than for unthinned stands. Another study (Olson and Chan 2005) found that wooded areas averaging as narrow as 15 ft could offset changes in microclimate that otherwise would occur as a result of thinning upland forests. As a result of a variable retention harvest in one forest, wind speeds increased threefold and consequently the loss of moisture from evapotranspiration also increased threefold (Bladon et al. 2003).

Other studies have looked more specifically at the *degree* of thinning or partial cutting that triggers changes in microclimate in leave-tree clusters and other small forest patches. A study of 1-hectare forest patches in western Washington (Heithecker and Halpern 2006, 2007, Aubrey et al. 2009) found that when only 15% of a forest canopy within a patch was retained, it did little to protect the remaining forest from microclimatic changes induced by adjoining clearcut areas. When canopy retention within the patch reached 40%, the light conditions were indistinguishable from those in an unthinned forest. Mean air temperature in those patches was significantly cooler than in clearcuts, but maximum air temperature did not differ. Mean and maximum soil temperatures did not differ among patches with intermediate levels of canopy retention, but soil temperatures varied temporally to a greater degree as thinning within the forest patches increased. Different levels of canopy retention had no detectable effect on *minimum* air and soil temperatures, or on late-summer soil moisture. For most biological responses, the total amount of canopy was more important than how it was distributed horizontally (clumped or dispersed). In another study, thinning to a density of 80 trees/acre within a stream buffer did not affect soil temperature in streamside areas or the water temperature in a stream (Olson and Chan 2005).

3.3.2 Effects on Wetland Plants

The species composition of trees and other vegetation that survives or colonizes a forested wetland following the harvesting of timber in or near the wetland depends on many factors. These include the species of trees that were harvested, the initial structure of the forest stands, other vegetation that was present pre-harvest, the intensity and pattern of harvest, elevation of harvest relative to post-harvest water table level, logging techniques and equipment used, woody debris management practices, replanting and weed control efforts, soils, climate, topography, and other factors. The post-harvest species composition is of obvious interest because it affects a site's future biodiversity, wildlife habitat suitability, microclimate, disease and fire susceptibility, soil development, nutrient cycling, water balance, and many other aspects of structure and function. This section (3.3.2) primarily describes the following effects on vegetation that are associated with timber harvesting:

- shifts in vegetation species richness and invasive species
- shifts in vegetation structure (horizontal and vertical) including the amount of dying, dead, and down wood

The effects of timber harvesting on microclimate were covered in the preceding section.

<u>Shifts in Plant Species Richness</u>

The species richness of **native** understory plants is usually an excellent indicator of the effects of forest practices, but is sensitive to survey methods and data processing procedures (Loya and Jules 2008). Within a few years following most types of harvest, richness of native herbaceous plants normally increases in the cut-over areas. This may be particularly true where alder colonizes harvested areas because alder tends to enhance soil fertility (Hanley 2005). However, after most harvests, herbaceous plant richness usually declines within the remaining forests, at least within 10 m from forest edges (Halpern and McKenzie 2001, Halpern et al. 2005). In Manitoba, total native plant diversity was approximately 30% higher on clearcuts than in woodlands, whereas moss and lichen diversity and cover were greater in wooded control sites (Locky and Bayley 2007).

Some native wetland plant species decline in response to more frequent and prolonged anaerobic and acidified conditions associated with the higher post-harvest water table, while others that are more adapted to aquatic conditions may increase as a result of increased duration of adequate soil moisture in wetlands that are (or were) inundated only temporarily or seasonally (Tiner 1991). Whether higher post-harvest water tables depress or enhance on-site plant species richness depends partly on the pre-harvest conditions and seasonal timing of post-harvest water surpluses. Wetland plant richness may increase most dramatically as soon as a decline begins in the logging-related higher water table levels, as has been observed in former beaver flowages.

Gradually, as trees (especially conifers) regenerate in cut-over areas and shade out shade-intolerant graminoids and forbs, understory plant diversity declines in harvested areas. But eventually, as trees approach late-successional stages and gaps form in the forest canopy, the diversity of native understory plants increases (Halpern and Spies 1995). Although probably no plant species in the PNW is associated with a single successional stage, a majority of forest plants are most abundant in old growth (Halpern and Spies 1995). As opposed to clearcutting, thinning sometimes accelerates this second burst of plant diversity, and may pose less risk of facilitating the spread of invasive plants. Green tree retention ("lifeboating") also appears to accelerate the post-harvest recovery in clearcuts of ectomycorrhizal fungi, epiphytic lichens, and small ground-dwelling animals, while perhaps being less effective for restoring the diversity of mosses and vascular plants (Rosenvald and Lohmus 2008).

As the area of a patch of natural habitat increases within a landscape, so does the diversity of native plant species. Blocks of forest smaller than about 9 acres may be less capable of supporting the expected array of mosses in British Columbia (Baldwin and Bradfield 2007). Forested wetlands and stream banks next to waterfalls contain greater abundance and taxonomic richness of lichens and mosses than do drier forests, at least on the windward slopes of inland rain forests of British Columbia (Radies et al. 2009). Contributing factors include the higher moisture levels, greater resistance to fire, and relatively mild temperatures. These conditions make these habitats refugia for rare lichens. A leveling off of the plant species-area accumulation curve in Alberta forests appeared at a forest patch size of about 27 acres (Gignac and Dale 2007). A study in Washington found that forest patches as small as 2.5 acres, if not narrow, may be large enough to have a microclimate supportive of most native plants and animals (Heithecker and Halpern 2007). In Wisconsin forests that were studied over a 55-year period, patches which were larger and contained more surrounding forest cover retained more plant species and were more likely to be colonized by new native species than smaller forests in more fragmented landscapes. Plant community composition was better explained by the amount of surrounding forest than by environmental factors within the studied forests (Rogers et al. 2009). The richness of native plant species specifically in wetlands has been found to correlate negatively with the extent of landscape alteration measured as far as 250-300 m from the wetlands (Houlahan et al. 2006). However, many factors such as differences in land use history are not well-accounted for by such correlational studies.

Shifts in Vegetation Composition and Structure

The most obvious harvest-related change in vegetation structure is a temporary loss of large trees in the harvested area and -- due to increased windthrow -- along logging roads and in adjoining forested wetlands. This results in a loss in the vertical complexity of habitat and also reduces the epiphytes (particular mosses and lichens) characteristically associated with larger trees, especially in wetlands. The other major structural effect is the accompanying increase in herbaceous plants and shrubs (e.g., MacCracken 2005), which has been shown to occur within forested wetlands even if the wetlands themselves remain uncut and separated from upland timber operations by a narrow buffer (Palik and Kastendick 2010).

After forested wetlands are logged, the time required to return the wetlands to approximately the same forested condition (with regard to canopy cover, species composition) usually is much greater than in recovering upland forests. This is particularly true if large parts of the wetland remain flooded for weeks or months at a time during the growing season, and/or if soils are relatively infertile, compacted, or peaty (Prescott et al. 1996). In such situations, restoration of forest cover depends partly on the extent of mounded microtopography, but the large root wads and residual logs that create such mounds are sometimes removed by logging equipment even during winter operations (MacKenzie and Moran 2004). Elevated mounds allow flood-sensitive tree seedlings to become established because they keep the seedlings out of the zone where prolonged inundation and soil anoxia can stymie growth. Site treatments intended to simulate natural mounds have generally not increased seedling survival in wetlands with well-humified, highly erodible organic materials (MacKenzie and Moran 2004). At least in conifer-dominated wetlands in western Canada, mounding can promote establishment of acidifying mosses between mounds which may reduce forest productivity in the longer term (Asada et al. 2004).

Within logged wetlands, poor tree regeneration means a delay in return to prior forested conditions. This in turn means that the consequently increased water table in the logged wetlands, with deeper and more seasonally persistent surface water, may continue for years longer than it takes for stream flows to return to pre-harvest conditions following the same operation. Depending on site characteristics, prolonged elevation of the water table can either increase or decrease the richness of herbaceous plants, but it nearly always hinders the growth and diversity of trees, especially on peat soils (Emil et al. 2006). In wetlands near Puget Sound, the height of 50-year-old trees (western red cedar, Sitka spruce, western hemlock) is only about 90 feet and this is approaching the maximum they are likely to reach in many wetlands in that area (Painter 2007, 2009). The same study found that those tree species in Puget Sound forested wetlands averaged 15-18 inches in diameter when mature, and 27 inches in a few old-growth forested wetlands.

Compared with conifers, mature hardwoods tend to produce higher densities of snags required by cavity-nesting wildlife, may export more nutrients to adjoining waters (Roberts and Bilby 2009), may be of greater overall importance to bird species known to be declining in the PNW (Betts et al. 2010), and are more suitable than conifers for wetland mammals such as beaver and moose. Beaver subsequently create wetlands along streams. Although beaver dams can temporarily restrict fish movements, the waters impounded by beaver dams are often more productive in terms of both the number and size of fish (Leidholt-Bruner et al. 1992, Schlosser 1995) and the variety of other plants and animals (See reviews by Naiman et al. 1988, Pollock et al. 1994, Stoffyn-Egli and Willison 2011).

Among shrubs, wetland-associated hardwoods (e.g., alder, ash, paper birch, aspen, cottonwood, willow, dogwood) tend to colonize many harvested sites more readily than conifers (e.g., western red-cedar, western hemlock, Sitka spruce, larch)(Villarin et al. 2009). Alder increases in response to increased soil moisture (caused by reduction of tree-associated transpiration) and increased light availability, as well as increased phosphorus availability (Uliassi et al. 2000, Brown and Courtin 2003, 2007). Red alder becomes dominant mainly on mineral soils, and tends to thrive better than most of the region's trees where conditions on these soils are more open and wet. Its positive response to logging is generally similar to its response to landslides and severe floods, and is particularly pronounced where two or three of these events have occurred at the same location within a few decades (Gomi et al. 2006). Red alder saplings fare poorly in closed-canopy forests, in clearcuts on upland sites (Haeussler et al. 1995), on sites heavily grazed by cattle or wild ungulates (Case and Kauffman 1997), or where soils remain water-covered for as little as one week during the growing season (Ewing 1999). Once mature tree height is reached, most red alder can survive low-intensity fire (Fonda 2001).

Because red alder quickly forms dense canopies on many sites, over a span of several decades it can inhibit the regeneration of many shade-intolerant woody and herbaceous species, thus reducing vertical habitat complexity in some instances (Hibbs and Giordano 1996). In British Columbia, the stem volume of western red-cedar was significantly less where red alder density exceeded 500 stems per hectare, whereas paper birch was not detrimental to western red-cedar growth until its densities exceeded 1000 stems per hectare (Cortini and Comeau 2008). Where red alder in southeast Alaska riparian forests comprised more than 53% of the total live basal area, the density of other tree species, their basal area, and mean diameter were significantly related to the percent of alder, measured as a proportion of total stand live basal area (Orlikowska et al. 2004). Conifer regeneration under dense alder canopies is influenced by distance from conifer seed sources, and perhaps slightly by understory cover (Beach and Halpern 2001).

For several years after wetlands are logged, vegetation structure may also shift as a result of a shift from mainly facultative wetland species to a higher proportion of obligate wetland species, caused by the logging-related increase in soil saturation noted previously. The specific types of structural and species composition shifts that will occur (as predicted by state-and-transition models or succession models, for example) have not been documented in forested wetlands of the PNW, but will likely depend on the plant species originally present; seed bank composition; magnitude and timing of post-logging water table changes; changes in light, sediment deposition, and nutrients; and proximity and connectivity to other wetlands. At least in Quebec, most wetland plant species are less limited by dispersal than are upland species, so fragmentation of wetlands by forestry operations might pose less threat of reducing wetland plant diversity than is the case for upland plants subjected to fragmentation of upland forests (Flinn et al. 2010).

Rotation length (the time between harvests of the same forested site) and fire suppression policies also influence vegetation structure and consequently habitat over long time periods. Short rotations (harvests that occur before trees reach much of their potential height) often deplete coarse woody debris and limit the extent of old-growth structure important to some wildlife species (Bunnell et al. 1997). Forest age and width of nonharvested buffers may be more important than the rotation age of plantation forests in supplying wood to streams and wetlands over the long term (Meleason et al. 2003). The effects on forested wetlands of rotation lengths in upland forests that adjoin wetlands have not been studied, nor has the structure of a series of "virgin" (or long-rotation) forested wetlands been quantified and compared with that of forested wetlands in similar geomorphic settings but harvested at shorter intervals and intensities.

Fire, although infrequent in most forested wetlands of the PNW, potentially affects soil organic matter (especially in bogs and other peat wetlands), biogeochemical cycling, hydrologic processes, vegetation composition and structure, amount of snags and down wood, and overall habitat quality. This has not been quantified, and policies that require suppression of all fires are likely to affect some wetlands in a variety of ways.

Invasive Exotic Plants

Another vegetation change that sometimes occurs in association with tree removal and construction of logging roads is an increase in non-native plant species. Non-native plants are a concern partly because they typically reduce the diversity of native plant species present at a site (Perkins and Willson 2005, Magee et al. 2008). Invasive plants tend to be more prevalent in narrower buffers and in smaller unshaded, recently-formed wetlands that experience large water level fluctuations. Some studies have inferred a reduction in productivity or diversity of wildlife populations where non-native plants have invaded, but no studies in the Pacific Northwest have yet demonstrated this. The assumption is well-founded, being based on the fact that the most common invasive plants typically simplify the physical structure and reduce the diversity of the plant community by outcompeting more diverse assemblages of native plant species. To varying degrees, native fish and wildlife species have come to depend on native plants (specifically their berries, seeds, foliage, associated insects, and leaf litter) being available for food and/or cover at specific times of year. Thus, any significant simplification of vegetation structure, or shifts in the seasonal timing of food availability (due to different maturation, flowering, or fruiting times of non-native plants, or different times of leaf-fall, leaf decay, and nutrient release) is likely to adversely impact many fish and wildlife species (Burghardt et al. 2009, Rodewald et al. 2010). However, the likely effects of non-native plants on wildlife are probably very species-specific and region-specific, with some invasive plants benefiting particular native species of wildlife and others being detrimental or neutral (e.g., Kennedy et al. 2009, Kapust et al. 2012). Invasive plants can also alter biogeochemical cycling rates and chronology (e.g., Urgenson et al. 2009) and can hypothetically change the water balance and duration of saturation in small isolated wetlands.

Even when invading non-native species do not substantially diminish a wetland's native biodiversity or appear to alter its functions, they often indicate the occurrence of physical or chemical conditions that are outside the normal range of variation, thus potentially constraining some important ecosystem functions (Parendes and Jones 2000). Non-natives invade forested wetlands most often following soil disturbance, such as from fire; the introduction of cattle; population increases of native herbivores following timber harvest; and/or increased disturbance from natural (e.g., landslides, floods, windthrow) or human-associated factors (e.g., logging, agriculture, construction of roads, trails, gravel pits) (Harrod and Reichard 2000). Because non-native plants tend to have broad environmental tolerances, many are good indicators of inconspicuous changes to underlying ecosystem processes and functions. Moreover, the areas they come to dominate often become more resistant to further change (Werner et al. 2002, Wigand 2003, Stohlgren et al. 2002), which has both benefits and costs in terms of habitat for other species.

Logging roads in particular are often the first point of entry for exotic species into a landscape, and roads can serve as corridors along which disturbance-associated plants move farther into the landscape, carried by logging equipment, humans, and animals that travel along road corridors (Lonsdale and Lane 1994, Greenberg et al. 1997). Many invasive plants occur in or near disturbed areas such as gravel pits, and seeds of these plants can be dispersed rapidly throughout a forest and into its wetlands when gravel is spread on roads.

Because most invasive species prefer upland to wetland environments, forest practices that lower wetland water tables increase the risks to forested wetlands from invasive plants. On the Olympic Peninsula, early-successional forested wetlands (e.g., alder flats) in floodplains were found to be more susceptible than uplands to invasion by non-native plants, and hosted even more invasive species than recent clearcuts (DeFerrari and Naiman 1994). Nonetheless these areas were not a major source of non-native plants to other habitats. The same study found that the presence of invasives was not predicted by patch size of the associated forest, slope, aspect, elevation, or to distance to human population centers, major highway, or river mouth. Ditches that accompany forest roads can serve as a vector for movement of invasive plants into wetlands (Watterson and Jones 2006). The indirect hydrologic, chemical, and biological effects on wetlands of non-native plants that often dominate clearcuts in the PNW but seldom invade wetlands (e.g., Scotch broom, Himalayan blackberry) have not been studied.

Most invasive plants in the PNW prefer unshaded conditions (Washington Department of Agriculture 2013). Nelson and Halpern (2005) found that non-native plants invaded forests and reduced native plant richness mostly within 10 m of the forest edge. A study of Alberta forests found that non-native plants were most abundant between 15 and 50 ft from the forest edge, and some species were found up to 130 ft from the edge. In hardwood forests of Wisconsin, forest roads increased the abundance of invasive plants and reduced native species richness within 5-15 m of the edges they created (Watkins et al. 2003). Although larger patches of forest generally support more non-native plant species than smaller fragments, the smallest fragments sometimes have the greatest number of non-native species per square meter (Gignac and Dale 2007). Dense vegetation tends to restrict wind-driven dispersal of seeds of many non-native plants into forests (Cadenasso and Pickett 2001). However, a study in North Carolina reported that riparian forests were not better protected from invasion by non-native plants by wide than by narrow buffers (Vidra and Shear 2008), perhaps emphasizing the importance of considering individual species traits and geography rather than relying on general paradigms.

3.3.3 Effects on Fish and Aquatic Invertebrates

Literature on the effects of forest practices on anadromous salmonid fish was reviewed by Liquori et al. (2008). Coho salmon and cutthroat trout in particular use both forested and herbaceous riverine wetlands extensively for rearing or overwintering (Heifetz et al. 1986, Brown and Hartman 1988, Swales and Levings 1989, Beechie et al. 1994, Henning et al. 2006). Some of these wetlands (usually termed off-channel areas, backwaters, oxbows, or floodplain wetlands) are connected to larger rivers only intermittently during annual high water, yet are heavily used, especially where they contain or adjoin deep pools. In one instance, stream reaches with wetlands accounted for less than 1% of the total available coho habitat in a watershed, yet supported over a third of all juvenile coho salmon (Murphy et al. 1989). Coho emigrating from streams with many wetlands are larger than those from higher velocity streams (Bennett 2006). Where stream temperatures are not near thresholds for coldwater fish, the streams which flow through meadows have sometimes been found to be more productive than either densely forested streams or logged streams with an open canopy (Dolloff 1987). Larger smolt size leads to increased overwinter and marine survival of juvenile anadromous salmon (Ebersole et al. 2006). Based on a full range of coho life history needs, the optimal size for an off-channel, salmon-accessible wetland or pond was estimated by one study to be 5,000–10,000 m^2 (Rosenfeld et al. 2008). In streams -- and perhaps also in some lakes (Roth et al. 2007) and wetlands -- large submerged wood provides fish with important cover, although even in streams, other factors are often more predictive of fish abundance (e.g., (Bjornn et al. 1991, Bugert et al. 1991). Shorter forest harvesting rotations could mean a gradually decreasing supply of large logs falling into streams and wetlands (Anderson and Meleason 2009). Where fish are present, these provide essential cover for fish over the long term.

If they contain fish at all, permanently inundated wetlands and ponds that never connect to rivers or lakes are occupied mainly by non-game species (e.g., sculpins, dace) and various introduced species (e.g., bass, brown trout). Other factors (such as elevation) being equal, temperatures in such wetlands tend to be warmer than in wetlands connected to channels, and suspended sediment from logging or other sources tends to be deposited and accumulate. Even where fish access to headwaters is limited, the carbon from headwater wetlands (and particularly from their herbaceous plants) has been shown to have a significant effect on downstream food chains important to anadromous fish at some locations (Dekar et al. 2012). Aquatic invertebrates that drift or are washed out of inaccessible headwater wetlands also benefit downstream fish populations.

Forest roads can impact fish use of wetlands by blocking access if not culverted, or if culverts are inadequately sized or placed. In one Washington watershed, 13% of the historical coho habitat was lost as a result of barriers created by improper culvert placement or design (Beechie et al. 1994). Timber harvest operations also can reduce water depth and persistence in wetlands as a result of increased sedimentation (see section 3.2.2). At least in theory, this can diminish the suitability of pools and off-channel wetlands as fish habitat (Bjornn et al. 1977, Reiser and Bjornn 1979, Jackson and Beschta 1984, Alexander and Hansen 1986). Timber harvest also can adversely affect anadromous fish by increasing water temperature (section 3.2.1). However, the temporarily increased water levels resulting from tree removal in or near wetlands (section 3.1) can, in theory, also increase access and habitat space for fish and aquatic invertebrates. The resulting increased connectivity of headwater wetlands to mainstream areas can temporarily facilitate not only fish movements, but also the export of carbon and aquatic insects from headwater wetlands to downstream food chains.

Many studies in the PNW have found that partial removal of a canopy that otherwise completely shades small streams can increase the growth and biomass of algae, aquatic invertebrates, and/or fish (Hawkins et al. 1983, Hetrick et al. 1998a, 1998b, Keith et al. 1998, Fuchs et al. 2003), at least until the riparian forest grows back. When this happens, invertebrate abundance temporarily increases but the number of taxa within the water body sometimes declines temporarily, especially if a substantial influx of fine sediments occurs (Richardson 2008). In British Columbia, the invertebrate biomass in recently harvested streams was nearly twice that in non-harvested or older harvested sites (Fuchs et al. 2003). No differences were detected among the 3 stream categories in regard to substrate composition, large organic debris density, or dimensions of pools and riffles. Also, temporary post-harvest shifts from conifer trees to hardwood shrubs (particularly alder) often occur, and could increase the amount of nutrients available to aquatic invertebrates and fish. In Southeast Alaska, streams adjoined by extensive alder have a greater abundance of invertebrates than those adjoined by young conifers (Wipfli 1997, Wipfli and Musslewhite 2004). This difference can be so great that, if downstream habitat is not limiting, the invertebrates could support a biomass of fish that is four times that of the conifer sites (Piccolo and Wipfli 2002).

In the Pacific Northwest, wetlands have apparently not been the focus of studies of the effects of timber harvest operations on invertebrates and fish. In aspen-dominated landscapes of Minnesota, invertebrate *richness* in seasonally ponded wetlands was found to be greater next to young than old (>35 year old) aspen stands, perhaps due to more light reaching wetlands next to open-canopied young stands, which triggered extensive growth of submerged macrophytes that support larger numbers and diversity of invertebrates. When old-growth aspen stands surrounding 16 wetlands were clearcut, the wetlands subsequently experienced longer duration of standing water, increased primary productivity, and both increases and decreases (depending on the functional group) in *abundance* of their invertebrates (Palik and Kastendick 2010, Hanson et al. 2009, 2010). Algae exhibited no significant response to harvest treatments or to the measured environmental variables. Another Minnesota study of timber harvesting effects on wetlands (Palik et al. 2001) found many physical characteristics of small seasonal ponds were unaffected by harvest of the adjacent upland forest. Canopy openness was one characteristic that was affected. Small wetlands with more open canopies had greater total macroinvertebrate abundance and a larger number of sensitive species. Whether these relationships would hold true for conifer-dominated landscapes and species in the PNW is unknown.

As noted in section 3.1, timber harvests located upslope from wetlands typically do have measurable effects on water levels downslope. Literature on ecological effects of streamflow conditions, which was reviewed by Poff and Zimmerman (2010) and Poff et al. (2010), may have some relevance for predicting ecological responses to changing water levels in wetlands, but effects are unlikely to be identical to those noted in streams because wetlands host different species and have different geochemical conditions.

In the Pacific Northwest, the contrast between the streamside and upslope forest is not as pronounced as that found in drier regions (Richardson et al. 2005a), and the same is likely true of the region's forested wetlands. In interior British Columbia, aquatic invertebrate communities were found to differ more between streams than between harvest treatments (Melody and Richardson 2007). In particular, some aquatic processes and species assemblages in streams are distinctly different in headwater areas compared with lowland portions of streams (Richardson and Danehy 2007). Some studies have found very different water quality in headwater (slope) wetlands and as compared with riverine wetlands at lower topographic positions (Cole et al. 1997a), whereas other studies have found either no differences or only minor differences (Azzolina et al. 2007).

In the Snohomish River Basin of western Washington, adult coho abundance and median adult coho densities were correlated at a landscape scale with wetland occurrence, as well as with local geology, stream gradient, and land use (Pess et al. 2002). This finding applied at both the watershed and stream reach scales. Forest–dominated areas had 1.5 to 3.5 times the densities of coho than densities found in rural, urban, and agricultural areas. Similarly, in Puget Sound lowland streams, the condition of invertebrate communities correlated positively with both the total amount of riparian vegetation and with the contiguity of riparian with upland vegetation (Shandas and Alberti 2009). However, in an area of British Columbia where 21% of a watershed was clearcut, no adverse effects on summer or winter condition or abundance of coastal cutthroat trout were detected (DeGroot et al. 2007). Also, no increases in stream temperatures were detected, but weather in the post-logging years was generally cooler, confounding the detection of logging effects.

Effects of timber harvesting on aquatic food chains were reviewed by Hanson et al. 2010 and in meta-analyses of prior literature by Marczak et al. (2007, 2010). In Oregon, results from one study suggested that the proportion of a watershed that is harvested and its proximity to a water body may wield greater influence on aquatic invertebrate diversity in that water body than the density of forest roads in the watershed (Fore and Karr 1996).

Using data from a probability sample of 167 headwater streams in forested watersheds of western Oregon, Herlihey et al. (2005) found no statistically significant relationship between history of timber harvests and the prevalence of disturbance-tolerant stream invertebrate communities (represented by an Index of Biotic Integrity, IBI). Another regional study of headwater streams where forest management (harvested-roaded vs. unharvested-unroaded) varied found significantly higher aquatic invertebrate density at harvested sites, while taxonomic richness and diversity did not differ (Medhurst et al. 2010). Based on review of many studies in the PNW, Richardson (2008) concluded that for the first 10-20 years post-harvest, harvesting near streams increases the production and abundance of aquatic invertebrates but in some cases temporarily reduces their diversity. However, in some cases, the effects of harvesting on aquatic invertebrate assemblage can be detected for at least 40 years post-harvest due to associated changes in the type of forest cover (Zhang et al. 2010). The effects are not necessarily adverse.

Increased soil saturation following tree removal could temporarily increase surface water in wetlands, and thus habitat space. At the same time, increased sediment from logging could temporarily reduce habitat space and populations of some invertebrate species in shallow wetlands. Sedimentation could also possibly expand wetland area where sediments are deposited at the transition between shallow and deepwater areas. Increased seasonal persistence of water in post-harvest landscapes could increase the proportion of wetland species such as dragonflies that require more time to develop from larvae to adults. Increased light and warmer water temperatures from tree removal could temporarily favor species that feed largely on algae, provided that post-logging turbidity levels in the water do not remain high. This might occur at the expense of species feeding mainly on leafy detritus.

3.3.4 Effects on Amphibians and Reptiles

Many amphibians (frogs and salamanders) and selected reptiles (mainly garter snakes and turtles) are obligate wetland species in the PNW. This section focuses on those that require calm waters (ponds and some wetlands) at any point during their life. Most studies of the effects of harvesting operations on amphibians in the PNW have focused on stream-breeding amphibians that use wetlands only sporadically, such as Coastal tailed frog and Coastal giant salamander (e.g., Richardson and Neill 1998, Wilkins and Peterson 2000, Stoddard and Hayes 2005, Pollett et al. 2010) or forest amphibians that apparently have little or no affinity for wetlands (e.g., western red-backed salamander). A fewer number of studies of forest practices have included amphibians that inhabit small rocky seeps during part of their life, such as torrent, Dunn’s, and Van Dyke’s salamanders (Corn and Bury 1989, McIntyre et al. 2006). An unknown portion of those habitats would be classified as jurisdictional wetlands. As will become evident from the discussions in sections below, an even fewer number of studies have addressed the effects of forest practices on wetland- and pond-breeding amphibians such as northern red-legged frog, Cascades frog, Pacific treefrog, western (boreal) toad, northwestern salamander, rough-skinned newt, and long-toed salamander. There also have been no studies in this region of timber harvest effects on the wetland-dependent western painted turtle.

Most of this region’s wetland-requiring species also require upland areas with undeveloped (but not necessarily forested) cover, often at considerable distances from the wetlands (Semlitsch and Bodie 2003, Harper et al. 2008). Exceptions may be the

Oregon spotted frog, and turtles that lay their eggs on land but whose hatchlings and adults spend extensive time in water.The amphibians in this assemblage lay eggs in water (attached to wetland vegetation) and the mature hatchlings and adults spend extensive time on land. During most of their time on land, these species generally exhibit only limited movements (e.g., <10 m) in moist ravines, underneath logs, in burrows, or in other generally moist cover. However, long periods of relative immobility are sporadically interrupted by significant movements over sometimes large distances (> 1 mile), which occur mostly during rain and at night.

In general, frogs and toads tend to prefer more open-canopied wetlands due to the dependence of their tadpoles on algae and biofilms, which thrive best under conditions of high light and warm temperatures. Their submerged eggs also may have specific light needs in order to deter losses due to fungal growth. In contrast, adult pond-breeding salamanders are mostly predatory and can flourish in somewhat more shaded waters (Earl et al. 2011). Thus, other factors being equal, frogs in the PNW -- with the exception of the Coastal tailed frog -- would be expected to be impacted less adversely than salamanders by reductions in shade from trees, and might even be affected positively. This is suggested by data from harvesting studies outside the PNW (DeMaynadier and Hunter 1995, Clawson et al. 1997, Palik et al. 2001, Skelly et al. 2005, Todd et al. 2009).

In mostly developed landscapes of western Washington, richness of pond-breeding amphibians was strongly associated with proportional coverage of forest within 1 km of wetland edges (Richter and Azous 2000, Hayes et al. 2008). Data on species not found in the PNW further suggest that many amphibians avoid clearcuts and, to a lesser degree, partial timber removals (e.g., Rothermel and Semlitsch 2002, Semlitsch et al. 2009, Strojny and Hunter 2012). Maintaining buffers and corridors of unlogged habitat between wetlands or streams (Perault and Lomolino 2000, Lomolino and Perault 2000, Olson et al. 2007) as well as retaining large woody debris (Aubry and Hall 1991, Patrick et al. 2006) might partly mitigate reductions in populations that could otherwise occur for some amphibian and small mammal species, but this remains untested. Also, maintaining a hydrologically-diverse array of wetlands within the usual 0.1-10 km range of wetland-breeding amphibian populations might reduce risks of there being insufficient breeding habitat during any year due to cyclic drought or flood conditions. This has not been tested, and one review of published studies indicated that if amphibians can be considered to have “metapopulations” (a group of spatially separated populations of the same species which interact at some level), that is likely true for only some species at some locations (Smith and Green 2005).

Small pools that are excavated in clearcuts and along logging roads, either to provide gravel for road surfacing or to help mitigate loss of wetlands, have been shown in other regions to benefit several amphibian species (Barry et al. 2008). However, to support amphibian reproduction, they must be situated in a manner relative to groundwater seeps that ensures they retain water through their usual breeding period (DeMaynadier and Hunter 1995, DiMauro and Hunter 2002). Such a strategy may be worth testing in the PNW, with an eye to ensuring that it does not also facilitate spread of the regionally exotic and predatory American bullfrogs and green frogs, both of which prefers open-canopied, warm, permanent pools.

If tree removal, as described in section 3.2.1, increases water yield during critical breeding seasons for amphibians and thus increases the extent, depth, and duration of surface water (to at least 4 months and preferably 8-9 months duration), then amphibian breeding habitat may benefit. Changes in amphibian populations could have measurable effects on nutrient balances. For example, one study of frogs and salamanders found they exported 6-12 times more nitrate from wetlands than is imported when they lay their eggs there (Seale 1980).

Although roads in general are known to be a barrier to pond-breeding amphibians, snakes and turtles (as reviewed by Fahrig and Rytwinski 2009), and amphibian abundance in wetlands was correlated negatively with road density in Ontario landscapes (Houlihan and Findlay 2003), it is unclear whether narrow unpaved roads with low traffic volumes, as is typical of forest roads in the PNW, are capable of restricting movements of these species. In North Carolina, even some narrow logging roads that had long been abandoned continued to impair movements and densities of salamanders (Semlitsch et al. 2007). That road effect appeared to extend about 115 ft into the adjoining woods on both sides of the road. However, another study (Marsh 2007) found that narrow forest roads closed to traffic did not hinder salamander movements, at least not as much as wider roads with vehicle access. In eastern Canada, fewer amphibian species were found in wetlands having a high proportion of roads, precommercial thinning, and hardwood forest in their surrounding landscape, especially within 180 - 200 m (Houlihan and Findley 2003, Jacobs and Houlihan 2011). Well-traveled roads near Ontario wetlands were associated with reduced amphibian abundance and this effect extended more than 250 m from roads (Eigenbrod et al. 2009).

Inadequate culverting of forest roads may hinder movements of some salamander species (Sagar 2004, Ward et al. 2008). However, in another study (Patrick et al 2010), culvert characteristics did not appear to influence culvert use by salamanders crossing roads. If new culverts allow fish to access previously fishless headwater wetlands, populations of some amphibian species (e.g., long-toed salamander) in those wetlands could be reduced.

It must be understood that finding an individual frog or pond-breeding salamander in terrestrial habitat at a particular distance from a wetland does not necessarily mean that particular individual is representative of the population, or that uniformly-wide wooded buffers extending to that point, and contiguous to the entire wetland perimeter, are needed to ensure survival of populations of the observed species. It remains unclear whether very wide (>100 m) buffers that completely surround wetlands, as suggested for wetlands in the southeastern United States by Semlitsch and Bodie (2003) and Semlitsch et al. (2009), are necessary to support populations of pond-breeding amphibians in the PNW. This uncertainty about buffer configuration may be greatest in less-settled areas where most of the surrounding land cover matrix is undeveloped regardless of whether or all the land immediately adjoining and surrounding a wetland is so. Uncertainty of extrapolating findings from distant regions is also due to differences in climate and species in the PNW. As noted by Cushman (2006), "The suggestion that forest cover in the landscape benefits amphibians may not apply to all species that are fully aquatic or that depend on nonforested upland habitat." Because extrapolating research results from one species and region to others risks misinterpretation, research is summarized here by individual wetland species, focusing on those in the PNW that appear to be the most dependent upon wetlands and ponds.

Northern Red-legged Frog

In Oregon, adult red-legged frogs were captured more frequently in riparian and wetland areas than in upslope habitats (Gomez and Anthony 1996). Streams wider than 3 m may be used much less than those narrower than 1.5 m (Chan-McLeod 2003). Contrary to expectations, a study in western Washington forests found that stream-dwelling red-legged frogs were negatively associated with coarse woody debris (Kelsey 2000). Buffers of 30 m width on both sides of streams that adjoined 2-year-old clearcuts appeared to adequately support movements of this species during the year they were surveyed. In the mostly unforested landscape of Oregon's Willamette Valley, consistent year-to-year occupancy of wetlands by red-legged frogs was more strongly predicted by the percentage of a wetland's perimeter containing trees within 5 m of "shore" than by percent forest cover within a 500 m buffer or by presence of predatory bullfrogs and non-native fish (Adams et al. 2011). Breeding sites vary in terms of water depth, degree of permanency, fish presence/absence, and associated species (Richter and Azous 1995, Adams 1999, Richter and Roughgarden 2005). Ponds with little or no emergent or submerged vegetation (i.e., >75% open water) are less likely to be used in consecutive years for breeding (Pearl et al. 2005, Adams et al. 2011). Wetlands preferred for breeding are sometimes very small. On Vancouver Island, 97% of wetlands identified by Beasley et al. (2000) as suitable for this species were <0.1 ha, and red-legged frogs were present in 26% of these wetlands. The same study found the species in a variety of wetland types -- shallow open water, marsh, swamp, fen, and bog wetlands -- but with the greatest frequency in bogs and fens. Hardwood (e.g., alder, willow) stands may be favored as nonbreeding habitat in some areas of Oregon (Gomez and Anthony 1996, Cole et al. 1997b) but data are inconclusive

Forests were identified as important to this species by Pearl et al. (2005) as well as by Hayes et al. (2008), who also noted its regular seasonal movements at distances ranging over 1000 m from wetlands. Studies in British Columbia (Chan-McCleod et al. 2000) demonstrated the ability of the species to travel >300 m. In Washington, the frogs were found 1.25 times more often in successional forests (30–76 yr) than in clearcuts (Bury and Corn 1988). In Oregon, Cole et al. (1997b) reported that logging did not significantly alter their capture rates compared to uncut controls, but Aubry (2000) reported they were found 5–10 times more often in pitfall traps in rotation age stands (50–70 years old) compared with younger age classes including clearcut sites. However, it is not apparent that either of these two studies accounted for possible differences in proximity of treatment sites to natal wetlands, and that is likely to have significantly confounded the study findings.

If clearcuts do pose a barrier to essential upland movements of this species, after 11 years of regeneration the barrier effect may be substantially reduced (Chan-McCleod 2003). In the Vancouver Island survey (Chan-McCleod et al. 2000), the species was present in 32% of 11 wetlands that were in logged and/or roaded areas and in 24% of 27 wetlands that were in unlogged old-growth. In another British Columbia study, 11 frogs were found in a 70-year-old second-growth stand before harvesting and only one frog one year post-harvesting (Maxcy 2000). However, the occurrence of this species among 85 wetlands in one part of western Oregon was not predicted by the amount of surrounding forest cover within 100 m or 1 km (Pearl et al. 2005). Breeding ponds favored in California were mostly unshaded by a forest canopy (Cary 2010).

Of particular note is a study that examined the effect on frog movements of residual (leave-tree) patches in clearcuts (Chan-McLeod and Moy 2007). That study found frogs moved towards large (>0.8 ha) patches and away from small (0.3 ha) patches 50 m away. If patches were spaced within 5-20 m of each other, frogs moved towards the forested patches rather than randomly. Frogs placed in small patches of trees were less likely to leave those patches if there was water nearby.

<u>Pacific Treefrog (Northern Pacific Chorus Frog)</u>

When not in larval stages, this species inhabits a wide variety of terrestrial habitats, including grassland, chaparral, woodland, forest, and farmland. Among wetlands, it also tolerates a wide range of hydroperiods (durations of standing water). In western Oregon, this abundant frog was found to occur most frequently in wetlands where there was the least amount of forest cover within 100 m and the least shade (Pearl et al. 2005). Studies in northern Idaho reached a similar conclusion (Goldberg and Waits 2009). Thus, this species might be expected to tolerate or benefit from tree removal associated with harvest. Individuals move up to several hundred meters between wetland breeding sites and nonbreeding upland habitats.

Western and Boreal Toads

In much of their geographic range, these closely-related species appear to prefer treeless areas or open-canopy forest. One study near Vancouver, BC (Deguise and Richardson 2009) determined that the movements of this species -- unlike those of the Northern red-legged frog -- do not appear to be restricted by clearcuts. In fact, toads appeared to move along forest roads more frequently than at random relative to the proportional area of those roads, and when placed in forests, they moved towards forest edges and clearcuts. They readily and rapidly colonize and reproduce successfully in created ponds and other ponds with mostly-unvegetated shallow water (Pearl and Bowerman 2006) and in the landscape devastated by the Mount St. Helens volcanic eruption (Karlstrom 1986, Dale et al. 2005). Non-breeding habitat of these species is seldom located next to their breeding ponds (Rittenhouse and Semlitsch 2007). In Idaho, toads spent almost 60% of their time in terrestrial areas farther than 10 m from the pond where they were born, which dried up late in the season. Individuals may travel up to 40 m daily, and seasonally they typically moved at least 0.36 (females) to 0.69 miles (males) from ponds, generally favoring shrublands and open forest (Muths 2003, Bartelt et al. 2004). However, in Alberta this species was more abundant in forest up to 100 m from lakes than in forest 400-1200 m away from the lake (MacDonald et al. 2006).

Northwestern Salamander

This species appears to favor deeper ponds and wetlands with shrubs and sedges that extensively overhang permanent water, or with extensive (>20%) submerged vegetation. They appear to especially favor ponds that lack predators such as fish and rough-skinned newts (Pearl et al. 2005, MacCracken 2007). Adults spend extensive time on land in concealed microhabitats, and tadpoles often overwinter within wetlands that contain water year-round. Most individuals appear to stay within a few hundred meters of their natal wetland. Effects of forestry practices have not been determined, but they have been found in both forested and clearcut areas (Grialou et al. 2000).

Rough-skinned Newt

In Oregon, adult newts were captured more frequently in riparian and wetland areas than in upslope habitats (Gomez and Anthony 1996) and were very common in alder stands (McComb et al. 1993, Cole et al. 1997b). Also in Oregon, this species was found most frequently in wetlands with the greatest extent of surrounding forest cover, measured within 1 km, and with the lowest road density within 1 km (Pearl et al. 2005). In contrast to most salamanders, newts commonly are active in unshaded terrestrial areas in daytime and sometimes move hundreds of meters between wetlands. In one study, harvest and herbicide applications appeared to have no detectable effects on capture rates of this species (Cole et al. 1997b) and they have been found in clearcuts (Grialou et al. 2000). However, neither study measured whether abundance of individuals differed between treatments, and results could have been confounded by differences in proximity to breeding wetlands (which were unreported). Similarly, after thinning treatments reduced Oregon conifer stands from 600 to 200 trees per hectare along streams, more newts were detected along stream reaches with thinned stands than along reference reaches; the streams had buffers of 6, 15, 70, and 145 m on each side (Olson and Chan 2005, Olson and Rugger 2007).

Long-toed Salamander

This pond-breeding species has been reported from a wide variety of habitats, e.g., semiarid sagebrush deserts, sub-alpine meadows, dry woodlands, humid forests, rocky shores of mountain lakes. Although forested areas were considered important to this species in northern Idaho (Goldberg and Waits 2009), dry moisture conditions in upland soils were found to be a greater barrier to movements than land cover (Goldberg and Waits 2010). An abundance of animal burrows and other microhabitats, such as talus slopes, that provide seasonal retreats or adequate moisture for the adults during their overland movements may be more important than the presence of forest, per se. In Alberta, adults were found in 3-year-old clearcuts as well as in 180-year-old forests, and occurred in areas that were being actively logged (Graham 1997). This species does not require permanently flooded wetlands for breeding. In fact, fishless wetlands that are inundated only temporarily each year may be favored (Pearl et al. 2005).

Dunn’s, Van Dyke’s, and Torrent Salamanders

Dunn’s salamander was found in a clearcut (Grialou et al. 2000) but no information is available regarding its productivity there or in other habitats affected by forest practices. Both Dunn’s and Van Dyke’s salamanders are typically found in headwaters or otherwise very close to streams (Corn and Bury 1989, McIntyre et al. 2006, O'Donnell et al. 2007). It is likely that some vegetated stream microhabitats used by torrent salamanders (*Rhyacotriton* spp.) qualify as wetlands, but there are no data on wetland use by this species.

3.3.5 Effects on Wetland Birds and Mammals

As is true of plants, animal species can be classified as wetland-obligate (found almost exclusively in wetlands), facultative-wet (found mostly in wetlands but regularly in uplands as well), facultative (about equally frequent in wetlands and uplands), facultative-upland (found mostly in uplands but regularly in wetlands as well), or upland-associated. Excluding fish, most vertebrate species in the PNW would be classified as facultative-upland or upland-associated. However, local circumstances -- such as a scarcity of a favored upland food, presence of different competitors or predators, or extreme weather conditions during a particular month or year -- can cause a shift in the usual wetland dependency category of individuals of a species during at least some of their life stages.

This report focuses mainly on species that, under typically circumstances over most of their geographic range, would be classified as wetland-obligate or facultative-wetland. Wetland-obligate species can be further classified as mainly water-associated (e.g., loons, grebes, ducks, geese, swans, sandpipers, gulls, belted kingfisher) or vegetation-associated (e.g., herons, rails, marsh wren, willow flycatcher, common yellowthroat, Lincoln's sparrow). Birds that are not wetland obligates but sometimes associate disproportionately with streams, lakes, and wetlands in the PNW include cedar waxwing, tree swallow, veery, warbling vireo, and yellow warbler. In some locations this category also includes Pacific wren, western flycatcher, black-throated gray warbler, red-naped sapsucker, western wood pewee, Swainson's thrush, MacGillivray's warbler, song sparrow, and black-headed grosbeak (Lehmkuhl et al. 2007).

Contrary to expectations, bird richness (per unit area) in eastern Washington was found to be less in forested riparian areas than in forested uplands, and densities of individuals were about the same (O'Connell et al. 2000). In western Washington, stream riparian areas and uplands were nearly the same in terms of richness and abundance of birds (Pearson and Manuwal 2001), a finding echoed by riparian studies in British Columbia (Shirley 2005) . In northern Minnesota, species composition of bird communities did not differ significantly between seasonal wetlands and nearby forest (Hanowski et al. 2006).

Among the better-known mammal species that associate disproportionately with wetlands or wetland edges in forested landscapes of the Pacific Northwest are beaver, muskrat, mink, river otter, raccoon, moose, and many bats. Many other mammals, such as jumping mice, shrews, and their predators, can reach high population densities in unflooded wetlands and/or in riparian areas that adjoin more permanently flooded wetlands and streams (Anthony et al. 1987, Doyle 1990). In British Columbia, activity levels of bats were more than 40 times greater in riparian than in upland areas, due to greater abundance of emerging aquatic insects, and were significantly greater where stand complexity and extent of forest edges was greater; bat activity levels were not correlated with forest stand age (Grindal and Brigham 1999, Grindal et al. 1999). Flying insects and many songbird species also tend to concentrate along water edges, especially in landscapes where most remaining forest is in riparian buffers (Whitaker et al. 2000).

With rare exceptions, the geographic and elevational ranges of wetland-obligate and facultative-wetland species do not overlap parts of the region where wetlands are adjoined most frequently by timber harvests. The most strongly wetland-dependent species, such as shorebirds and waterfowl, occur in lowland landscapes dominated by agriculture or rangeland rather than forest cover. Some of these species might even benefit from harvesting trees in or surrounding wetlands in this region, where those trees hinder detection of predators or serve as perches for predatory birds. Also, if tree removal increases water yield and increases the extent, depth, and duration of surface water in wetlands during critical seasons (or temporarily creates new wetlands as described in section 3.1), then waterbird breeding habitat may benefit.

Dozens of studies in the PNW have examined the response of songbirds, and to a lesserextent mammals, to tree removal or tree cover generally. However, few if any studies have attempted to identify the sensitivity -- to various degrees and patterns of tree removal and forest roads -- of birds and mammals that associate strongly with the region's wetlands. None of the species that are commonly termed "forest interior specialists" – ones that have been found to occur mainly in unfragmented forests or wider buffers – have been shown to depend strongly on wetlands in this region.

As would be expected, immediately after clearcutting all forest-dwelling species disappear from the clearcut area unless some residual trees are left, and even then, not all formerly-present species will persist. Recent clearcuts are particularly devoid of birds in winter, but silvicultural treatments such as two-story and small-patch group-selection can lessen this effect somewhat, especially when sufficient snags, logs, and large trees are retained (Zarnowitz and Manuwal 1985, Chambers and McComb 1997, Steel et al. 1999, Zielke et al. 2008). Within a few years post-harvest, logging commonly results in greater shrub cover. Consequently, some wetland-associated songbird species that nest mainly in shrubs – such as willow flycatcher – respond positively in the short term to timber harvests (Chambers et al. 1999).

In British Columbia, Lance and Phinney (2001)compared nesting bird abundance and richness in 2 clearcuts, 2 partial retention sites, and 2 unlogged forest sites. Both bird metrics indicated high similarity between partial retention sites (15-22% of the stand was retained) and unharvested forest over a 3-year period (). The partial retention sites contained species from the forest, species from the clearcuts, and species found at neither of the other two site types. Some forest-dwelling species were missing from the partial retention sites, and some were less abundant, but others were more abundant at the partial retention sites. Another comparison study in Minnesota focused specifically on these treatments as applied near wetlands (Hanowski et al. 2006). A 17-m wide buffer was retained around the wetlands in all cases. Bird species composition in the partial retention sites (basal area reduced to 7-10 m^2 per hectare) was found to be more similar to uncut sites than to the clearcut sites. Another study by the same authors (Hanowski et al. 2005) found that when trees within riparian buffers were thinned to a basal area average of 17–25 sq. ft/acre, the number and variety of sensitive forest interior bird species declined in those buffers. The density of vegetation (e.g., basal area or percent canopy closure) in a buffer, corridor, or patch -- or in the landscape generally -- also influences habitat value for some species, perhaps as much or more than buffer width, corridor width, or patch size. In the Seattle metro area, the variety of breeding birds declined as forest canopy closure increased over the range of 45% to 100% (Donnelly and Marzluff 2006). That study found greater retention of native breeding birds where forests retained a tree density of at least 25 trees per acre.

Over the longer term, forest management practices in the PNW have favored conifers at the expense of deciduous trees. This may be a potential concern because a recent analysis of 42 years of PNW data on 12 songbird species indicated that birds associated with deciduous forest (chiefly maple, alder, oak) have declined at the greatest rates in this region (Betts et al. 2010). None of the studied species is strongly associated with wetlands, but many of the floodplain wetlands in this region have a large deciduous tree component important to birds generally (Lock and Naiman 1998). Depending on the species and scale of measurement (distance of 500 ft or 1640-6560 ft around nest site), between 1.35% and 24.5% cover of deciduous trees should be supported to sustain particular songbirds with declining populations in this region (Betts et al. 2010). Deciduous trees (particularly aspen, cottonwood, birch, and willow) are also important to American beaver and moose. These mammals often become established in landscapes disturbed by harvest or fire. In fact, one study in Ontario (Landriault et al. 2009) found 73% of beaver lodges adjacent to shorelines with clearcuts remained active, whereas only 34% of lodges with no shoreline clearcuts were active. The best predictor of lodge activity was the presence of a 21-35-year-old shoreline clearcut. That may have been the case because deciduous vegetation preferred by beaver reaches maximum density in clearcuts of approximately that age. However, another wetland-associated species -- mountain beaver – was found by Gyug (2000) to be sensitive to harvest activities that disturb soils, due partly to its habit of digging tunnels and dens in fine-textured soils that are subject to compaction from heavy machinery. (

Effects of forest roads on birds and mammals in wetlands have apparently not been studied in this region. Inadequately-culverted forest roads that block drainageways and thus sometimes create wetlands could increase habitat for waterbirds, wetland-associated songbirds, and wetland-associated mammals. Tree mortality resulting from an elevated water table on the uphill side of some unculverted forest roads could temporarily increase availability of snags for cavity-nesting wildlife, as could damage to uncut trees from increased windthrow on the edges of clearcuts, in small residual tree stands, and along forest roads. Windthrow or other sources of snags and down wood are a huge benefit to many wildlife species and support greater species richness of birds (Zarnowitz and Manuwal 1985, Żmihorski 2010). Thus, salvage harvesting can be detrimental to many species (Lindenmayer et al. 2008).

On the other hand, and at least historically, many small wetlands were permanently filled during the construction of forest roads, which commonly followed streams and lowlands where wetlands mostly occur (Pearson 2010). New forest roads can increase disturbance of a few wetland species that are highly sensitive to human intrusion during critical breeding periods (e.g., great blue heron, some raptors) as well as result in increased poaching and mortality from vehicle collisions. These impacts can be minimized to some extent by closing roads or restricting vehicle access during critical seasonal periods. However, for small mammals, the simple presence of a road, regardless of whether it is paved or the amount of traffic on it, has been shown to pose a significant barrier to movements. In one study in which animals were translocated, each road that chipmunks and mice had to cross reduced by half the number of individuals that crossed it (McGregor et al. 2008). Road widths varied from 6.8 to 14.4 m for the pavement width only, and from 6.8 to 17.8 m for the pavement plus gravel shoulders. Mean canopy cover ranged from 20 to 55 percent.

Foraging movements of some forest birds (none of them rare or known to be wetland-dependent) also can be constrained by large canopy gaps, perhaps including some of those caused by forest roads or timber harvests (Rail et al. 1997, St. Clair et al. 1998, Belisle et al. 2001, Belisle and Desrochers 2002, Shirley 2006, Tremblay and St. Clair 2009). Elk appear to avoid large areas near often-traveled forest roads (Rowland et al. 2000). Forest roads are expected to facilitate access of nest predators (e.g., ravens, skunks, squirrels) and nest parasites (brown-headed cowbirds) to nests of wetland birds, but long-term changes in local populations of prey or host species as a result of this have not been investigated in the PNW. A wetland study in coastal British

Columbia found that nest parasitism of song sparrow nests by brown-headed cowbirds was a significant drain on the local sparrow population (Rogers et al. 1997). Although that study did not specifically examine the role of fragmentation of forests by harvest, the association of both cowbirds and song sparrows with forest openings has been well documented. However, in central Pennsylvania, predation of forest bird nests was significant around clearcuts but not along forest roads, and predators did not appear to follow forest roads (Yahner and Mahan 1997). Edge effects (positive and negative) on terrestrial songbirds from timber harvests and roads, although perhaps relatively minor, appear to be greater in the more productive western parts of the PNW than in forests farther east (McWethey et al. 2009).

3.3.7 Effectiveness of Buffers for Maintaining Microclimate and Habitat Structure

Vegetated buffers, simply by adding structure that is more vertically and horizontally complex than that of recently logged areas, provide habitat to large numbers of terrestrial (upland) species. Buffers also can be designed to connect patches of important habitats, although for typically non-linear habitats such as wetlands, this happens more often by chance than design.

This section focuses mainly on the role of buffers in protecting those species that are most dependent on or strongly associated with *wetlands*, particularly in the Pacific Northwest. These include many plants, aquatic invertebrates, amphibians, and waterbirds, well as some fish and some terrestrial birds and mammals. Apart from maintaining wetland structure, it is also important to protect these species from detrimental changes in wetland microclimate, habitat fragmentation, and water quality degradation. Harvest impacts to microclimate are covered in section 3.3.1 and to water quality in sections 3.2.2 and 3.2.3.

Buffers to Protect Microclimate and Wetland Vegetation

Common objectives for wetland buffers for protecting wetland vegetation are:

- minimize the spread of invasive plant species into wetlands,
- maintain natural rates of windthrow of any wetland trees, and
- avoid impacts, especially to rare wetland plant species and communities, from altered microclimate and from increased sediment and nutrient inputs.

Apparently no studies have examined effects of harvest on wetland plants in the PNW, or the degree to which buffers minimize any undesired changes to wetland plant communities. In a Minnesota study, buffers 15 m wide surrounding seasonal wetlands lessened the effects of varied intensities of timber harvest in adjoining uplands, but did not eliminate the effects entirely (Palik and Kastendick 2010). With increasing canopy openness, there was an increase in the cover of sedges, grasses, willow, alder, and poplar. Along streams in western Washington that were surveyed 2 years after adjoining uplands were harvested, buffers of 15 and 30 m had more alder and greater cover of berry-producing shrubs, compared with control sites where adjoining upland forest had not been cut (Kelsey and West 2000a). Narrower buffers in British Columbia had more species of shrubs compared to wider buffers, but deciduous tree density was greater in wider buffers (Shirley 2004).

Logging of uplands next to wetlands accelerates windthrow rates in buffers and/or in unbuffered forested wetlands. This can reduce or nullify some functions of the buffer or the wetland. When trees are uprooted by wind, this stirs the subsurface soil, enhances microtopography (consequently diversifying the variety of soil soil moisture durations within the site), and potentially releases nutrients and exposes dormant seed banks. These can all diversify the wetland flora. However, increases in nutrient availability following windthrow are not inevitable (Keenan et al. 1994), and loss of trees can profoundly change the wetland microclimate over the long term, and over the short term if salvage logging is implemented. Along streams, wooded buffers of 75 ft width may be adequate in some cases to protect remaining trees from windthrow, according to a literature review by Pollock and Kennard (1998). When narrower than ~50 ft, buffers are more prone to wind damage (Lopez et al. 2006, Martin and Grotefendt 2007, Anderson and Meleason 2009). Amounts of downed wood measured from a wind-swept lakeshore up to at least 130 ft into the lakeside forest were greater than in forests farther from the lake (Harper and MacDonald 2001). In Oregon, the amount of downed wood in riparian buffers was unaffected by thinning operations in the adjoining upland forest unless the buffer was narrower than about 50 ft (Anderson and Meleason 2009). However, in California, researchers found that tree fall rates were abnormally high for a distance of at least 200 m from clearcut edges (Reid and Hilton 1998). One study found that within riparian buffers that adjoined clearcuts in Washington, tree fall rates were 26 times higher than normal for 3 years post-harvest, and may have caused the eventual replacement of conifers with deciduous hardwoods. A study in Southeast Alaska reported that cumulative stand mortality in the part of a stream buffer closest to a clearcut (i.e., 10-20 m from a stream) was more than double what occurred in uncut reference units. Future potential supply of large wood to the stream was diminished by 10% compared with unlogged areas (Martin and Grotefendt 2007). One survey reported that 11% of the harvest units experienced significant windthrow in their stream buffers (Martin and Grotefendt 2007). The amount of windthrow depends not only on buffer width, but on orientation of the cleared edge relative to wind, edge contrast (size differential of vegetation), the size of nearby clearings, tree species and age distribution, soil depth, and local topography (Ewers et al. 2007, Laurance and Curran 2008).

Buffers to Protect Aquatic Invertebrates and Fish

Buffers to protect naturally occurring assemblages of aquatic invertebrates and fish focus on protecting water quality and habitat structure needed by these taxa. Studies of the effectiveness of buffers for minimizing aquatic biological impacts of timber harvesting were reviewed by Hanson et al. 2010 and in meta-analyses of prior literature by Marczak et al. (2007, 2010).

Nearly all studies of buffer importance to aquatic invertebrates and fish have been conducted along streams rather than wetlands. The aquatic fauna of most wetlands, especially isolated wetlands, is quite different from that of streams. This is partly because most wetlands have less dissolved oxygen in their water and soil/sediment. Also, in non-riverine wetlands that lack inlet channels, nutrients and large woody materials are not replenished from upstream sources. Accumulations of large woody debris, whose continued long-term supply from riparian buffers can be important to aquatic invertebrates and fish in flowing waters, might be less critical to maintaining aquatic invertebrates and fish in wetlands because current velocities and scour risks are usually less. Cover can be provided in wetlands by structure-providing aquatic plants that normally occur more sparsely in flowing waters. Also, many wetlands contain surface water only briefly (if at all) during the growing season. Among wetlands that have limited surface water exchange and/or prolonged periods without surface water, the aquatic life would be expected to be more sensitive to (and/or adapted and resilient to) changes in shade, nutrients, and contaminants.

In a study of Minnesota wetlands, the retention of 15 m-wide forested buffers appeared to partially lessen the biological influence of logging further upslope (Miller 2001, Hanson et al. 2010). In Washington's Coast Range, following logging with buffers of less than 10 m on both sides of streams, no stream invertebrate group declined in the 3 summers following harvest, and sediment particle sizes differed little from reference streams despite extensive windthrow in buffers (Jackson et al. 2007). The influence of buffers on stream invertebrates in Ontario was found to be small compared to the influence of forest cover overall, measured at stream reach and watershed scales (Stephenson and Morin 2009). A study of 13 headwater streams in British Columbia found that uncut buffers of 30 m or more on both sides of the stream were needed to limit changes to invertebrates and algae, following clearcutting farther upslope (Kiffney et al. 2003). Tree buffers of 30 m provided more organic matter to streams than did buffers of 10 m for at least 8 years post-harvest (Kiffney and Richardson 2010). Streams with either buffer width received nearly as much organic matter as streams flowing through fully forested areas, although the organic matter from clearcut areas had a greater deciduous component, with consequent implications for the timing and amount of nutrients reaching the invertebrate communities in receiving waters.

Buffers to Protect Wetland Amphibians

Buffers for amphibians are intended to protect wetland microclimate, water quality, and appropriate habitat structure within the adjoined wetland. As explained in section 3.3.4, buffers also are intended to provide part of the suitable upland habitat which pond- or wetland-breeding amphibians require during their life cycle. To meet that need, some biologists have recommended wetland buffers of 538 ft (Semlitsch 1998) or even more than 1000 ft (Semlitsch and Bodie 2003). To date, nearly all studies that suggest a need for such large buffers are from streams (seldom from wetlands) in the eastern United States,. However, although there have been many fewer studies in the PNW, preliminary evidence (e.g., Hayes et al. 2008) suggests that large buffers that at least partially surround PNW wetlands will help ensure the survival of a few of this region's wetland-dependent amphibians. Likewise, the several studies that have been conducted of amphibian response to logging in the Pacific Northwest have focused only on stream-dwelling species (e.g., tailed frog, Pacific giant salamander) or on mainly terrestrial amphibians, e.g., Vesely and McComb 2002, Dupuis et al. 1995, Hawkes and Gregory 2012). Authors of a study of pond-breeding amphibians in Connecticut (Skelly et al. 2005) concluded, "conservation strategies dependent on universally applied, inviolate shoreline vegetation buffers may inadvertently contribute to species loss because species differ in their sensitivity to changes in canopy." And as noted by Cushman (2006), "The suggestion that *forest* cover in the [buffer] landscape benefits amphibians may not apply to all species that are fully aquatic or that depend on nonforested upland habitat." Nearly all the studies of amphibians and buffers published to date focus mainly on buffers near clearcuts (as opposed to partial cuts), and do not fully describe other connectivity factors that could be important to amphibians in the landscapes in which the harvests are located, e.g., density and pattern of logging roads, buffer slope, soil moisture, and ground cover.

Although buffers of 150 ft have been recommended along some Oregon streams (Vesely and McComb 2002), that recommendation was based mainly on needs of frog and salamander species that do not occur commonly in wetlands. In Maine, amphibians (none which occur in the Pacific Northwest) were more abundant in 35-50 ft forested riparian buffers than in adjacent clearcuts two years following harvest, indicating at least some benefit to retaining buffers of this width (Perkins and Hunter 2006). A meta-analysis of 31 studies of amphibian use of riparian buffers (Marczak et al. 2010) suggested that amphibian abundance may be less in buffers than in unharvested areas, and that abundance was unrelated to buffer width. Only two of the studies in that analysis involved pond-breeding species found in the Pacific Northwest. In the Washington Cascades, amphibians in wetlands monitored for 4 years appeared to be unaffected by uneven-aged harvests regardless of buffer width (from 0 to 61 m) around the wetlands (MacCracken 2005). The main species were Northwestern salamander and rough-skinned newt.

Of probably greater importance than buffer width is the suitability (quality) of the habitat within a buffer (Harper et al. 2008). During their upland phase, most amphibians appear to require extensive ground cover, large logs, burrows, and other places that maintain a moist microclimate even during dry periods (Kluber et al. 2008). Clearcuts not subjected to extensive slashburning and woody debris removal might partially provide that during the few years post-harvest. Thinned forests, windthrow areas, and regenerating clearcuts might also provide it, but data are lacking for the wetland species in this region.

Buffers to Protect Waterbirds

For purposes of this review, waterbirds include ducks, geese, swans, loons, grebes, herons, egrets, rails, bitterns, cormorants, sandpipers, plovers, and others that spend nearly all of their time foraging in or along standing water. For waterbirds, wetland buffers are intended to protect wetland water quality and, depending on the locally occurring species, to maintain habitat structure next to the wetland. In some cases, the buffers around wetlands are intended to hinder access of predators to nests of waterbirds (although their effectiveness for this has apparently never been tested), as well as limit disturbance from recreationists, livestock, and pets.

Most of this region's waterbirds seem not to need buffers that are forested, as suggested partly by their frequent use of wetlands amid agricultural lands (Hirst and Easthope 1981, Lovvorn and Baldwin 1996, Shepherd and Lank 2004, Slater 2004). For those species, trees that comprise a significant portion of the perimeter of small (<10 acre) wetlands sometimes discourage wetland use by swans, geese, and shorebirds, and it is surmised that is because those trees make convenient perches for avian predators (Shepherd and Lank 2004).

However, a few of this region's waterbirds nest occasionally or exclusively in trees, particularly trees that are of large diameter and/or very tall. An untested assumption is that the closer such trees are to wetlands or large areas of standing water, the more important they are to these species. In the PNW, these species include wood duck, common and Barrow's goldeneye, bufflehead, double-crested cormorant, green heron, and great blue heron. The first four of these species require nest cavities excavated previously by woodpeckers, particularly the big cavities created by pileated woodpeckers only in large-diameter trees (Aubry 2002, Hartwig et al. 2002). Thus, managing forests for long rotations and/or maintaining suitable large-diameter trees that attract and are used by pileated woodpeckers (which are not a wetland-dependent species) is essential to sustaining the four cavity-nesting ducks that *are* wetland obligates in the PNW. In British Columbia, nest cavities of Barrow's goldeneye averaged 90 m from permanent water (Evans et al. 2002). In northern Minnesota, wood duck nests averaged 80 m from water (Gilmer et al. 1978). These distances cannot be used to define preferred buffer widths around ponds, streams, and wetlands because it is unclear whether areas closer to those water bodies would have been used if more suitable cavity trees had been present closer. In fact, a few studies have found that nests in tree cavities or artificial nest boxes closer to water or over water were less successful (Richardson and Knapton 1993, Robb and Bookhout 1995). Other requirements of these ducks are detailed in a literature synthesis by Maggiulli (2009), who also created a Bayesian Belief Network (BBN) model based on the synthesis. Her report provides numeric definitions of high, moderate, low, and inadequate conditions for these ducks, as defined by specific combinations of tree diameter class and tree type (conifer/deciduous), snag density, canopy closure, and distance of these features to wetlands larger than 1.5 hectare, rivers, or fish-bearing streams.

When logging equipment or humans on foot approach individuals of some wildlife species, those species sometimes abandon their young or at least flee (termed "flushing") or interrupt their activities. When these disturbances occur regularly over periods of time, it reduces the food intake and weight of both adults and young. If chronic, that disturbance can eventually make their populations less competitive or put individuals at greater risk of predation or disease. The energy balance may be especially delicate for bird species that characteristically have annual migrations spanning multiple continents, such as many of the warbler and flycatcher species that are Neotropical migrants. However, it is the larger waterbird species (herons, swans) and perching hawks and eagles which tend to be the most wary, especially during nesting periods. Distances at which birds will or will not be disturbed by humans or noise vary depending on species, habitat, time of year, flock size, amount of visual screening by vegetation, and other factors (Korschgen and Dahlgren 1992). A wetland buffer of 30-100 m might be required to reduce disturbance of some waterbirds, and a few species sometimes take flight when humans or pets approach from as far as 200 m away. However, as long as they are not shot at, over time individuals of many waterbird species habituate to the presence of humans. A wetland buffer of dense shrubs, regardless of its width, can minimize disturbance of waterbird species that regularly inhabit wetlands such as shorebirds, herons, and cranes. To protect nesting waterbirds from disturbance, upland buffers appear to be necessary only for wetlands with extensive open water, as only those wetlands are used regularly by the larger nesting waterbirds.

Buffers to Protect Other Birds and Mammals in Wetlands

A meta-analysis of studies from many regions (285 data points) indicated that forest interior bird species overall do not necessarily prefer natural forest more than buffers, and wider buffers do not necessarily result in greater similarity between reference forest and buffer sites (Marczak et al. 2010). A meta-analysis by Price and McLennan (2002) reached similar conclusions. These analyses of bird responses originate mainly from studies of buffers along streams rather than wetlands. Also, data on mammal use of forested buffers are very limited. A majority of the studies published to date focus mainly on buffers near clearcuts as opposed to partial cuts. Also, most do not describe other connectivity factors that could be important to mammals and birds in the landscapes in which the harvests are located, e.g., density and pattern of logging roads, vegetation age and diversity, ground cover, slope aspect, clearcut width. Width of remaining forest patches has been shown to have much less effect on bird species composition in those patches than the overall proportion of a landscape that is forested (Rodewald and Bakermans 2006). For example, in southern Michigan, a study of forested depressional wetlands found that richness and abundance of songbird species which were considered generally wetland-dependent there were influenced less by wetland characteristics than by characteristics of the surrounding upland forest, e.g., total patch size, tree height diversity (Riffell et al. 2006).

In the PNW, a few studies have highlighted particular species that might be sensitive to buffer width and/or patch size in this region, but apparently none are wetland obligates. Brown creeper, golden-crowned kinglet, black-throated gray warbler, and Wilson’s warbler in western Washington used buffers less than intact forests, and/or used 30-m buffers more than 14-m buffers (Kelsey and West 2000b). In terms of species turnover rates, buffers narrower than 14 m (on each side of a stream) did not maintain the pre-logging upland bird community, and black-throated gray warbler was not found in buffers narrower than 30 m.

Surveys in the forested landscape of the Cedar River watershed east of Seattle also found that golden-crowned kinglet, brown creeper, and black-throated gray warbler were more common in wider buffers or in uncut forest, and were more consistently present between years in those wider buffers. Riparian buffer widths of ~150 ft were needed for these species in order to attain equivalence with numbers found in unharvested areas, and occurrence of most other songbird species was associated with buffers of 100 ft but not 50 ft on each side of streams (Pearson and Manuwal 2001).

A study in the Oregon Coast Range that compared buffers of 0 to 75 m width on each side of streams found that brown creeper, hairy woodpecker, and chestnut-backed chickadee were more likely to be present in wider streamside buffers (Hagar 1999). Even the widest buffers (131-230 ft) failed to support Hammond’s flycatcher, varied thrush, and golden-crowned kinglet. In another Oregon study, some of the nesting species whose numbers declined in stream buffers after logging were Pacific wren, brown creeper, golden-crowned kinglet, Wilson’s warbler, and Swainson’s thrush (Chambers et al. 1999). Birds associated with wider stream buffers in the Portland metropolitan area were Pacific wren, brown creeper, and Pacific-slope flycatcher (Hennings and Edge 2003). Based on bird data from the Oregon Cascades, buffers of 200 ft on each side of streams were deemed adequate to support corridor and refuge functions for birds in clearcut areas (Lehmkuhl et al. 2007).

A study in British Columbia compared stream buffer widths of 46, 121, and 230 ft, and found that Pacific (winter) wren, as well as golden-crowned kinglet, Townsend’s warbler, Hammond’s flycatcher, and varied thrush were detected more often in wider buffers (Kinley and Newhouse 1997). Also in British Columbia, stream buffers of ~150 m failed to support several species at densities equivalent to those in extensive uncut forests: brown creeper, pileated woodpecker, golden-crowned kinglet, varied thrush, and red-breasted sapsucker. However, at least 2 species -- warbling vireo and Swainson’s thrush – were more common in buffers than in uncut forest. Pacific-slope flycatcher was mostly absent from streamside buffers narrower than ~50 m in logged watersheds (Shirley and Smith 2005). In clearcut areas in Southeast Alaska, the species peaked in riparian buffers of 250 m (Kissling and Garton 2008). That study also reported that brown creeper and hairy woodpecker might be sensitive to buffer width. The study found few other nesting songbirds whose presence was correlated with buffer width, but it did not examine buffers narrower than 100 m.

In Quebec, golden-crowned kinglet, and Swainson's thrush were seldom found in buffers narrower than 65 ft (Darveau et al. 1995). In Alberta, the narrowest riparian buffers in which several species nested were as follows (Hannon et al. 2002):

66 ft = yellow warbler, song sparrow, black-capped chickadee, western wood-pewee, yellow-rumped warbler, dark-eyed junco;
328 ft = Swainson’s thrush, common yellowthroat, hairy woodpecker, brown creeper;
656 ft = western tanager, purple finch.

In the same Alberta study, forest-dependent bird species declined as buffer width narrowed from 200 to 100 m and narrower, and the authors concluded that 200 m wide buffers would be necessary to conserve all forest-dependent species from the pre-harvest bird community.

Most data on bird use of buffers come from streams and from regions other than the Pacific Northwest. Because those situations reflect very different faunas, those results cannot be validly extrapolated to this region. As explained by Schmiegelow et al. (1997) and Schmiegelow and Monkkonen (2002):

> "the magnitude of the fragmentation effects we documented is small compared with those observed elsewhere. Birds breeding in the boreal forest, where frequent small- and large-scale natural disturbances have occurred historically, may be more resilient to human-induced habitat changes, such as those caused by limited forest harvesting. "

And Tewksbury and Martin (1998) comment:

> "the effects of fragmentation are dependent on the habitat structure, the landscape context, the predator community, and the impact of parasitism. All of these factors may differ substantially in western ecosystems when compared to previously studied forests, making generalizations about the effect of fragmentation difficult."

Based on their extensive data from Vancouver Island, Schieck et al. (1995) concluded:

> "Most species of birds that occur in the Pacific Northwest may be less susceptible [than Eastern species] to adverse effects of forest fragmentation."

And finally, in a synthesis on this topic, Kremsater and Bunnell (1999) observed:

> "In the east and midwest many studies document increased predation and parasitism near edges; in the Pacific Northwest researchers have found little effect of patch area or negative edge effects"

Some studies have reported more *individual birds per unit area* in wider buffers (e.g., Kinley and Newhouse 1997, in British Columbia). In fact, most studies of birds and insects have found higher densities of birds in buffers than in areas of comparable size in the middle of extensive forests (Marczak et al. 2010). This could be because most buffer studies have been of buffers next to clearcuts containing unsuitable habitat, forcing birds with no other choice than to use remaining buffers or else move longer distances to find large forest patches. Several studies (e.g., Darveau et al. 1995, Hannon et al. 2002, Hanowski et al. 2006, Betts et al. 2006) have noted how clearcutting can crowd individual birds (presumably those that formerly nested in the now-clearcut forest) into remaining patches of forest (e.g., buffers) for at least 1-3 years post-harvest.

Also, among buffers of different widths, wider buffers are more likely to have higher bird and insect densities simply because of the greater likelihood that they possess a variety of vegetation types and size classes, which generally leads to higher avian abundance (Marczak et al. 2010). Also, wider riparian buffers in British Columbia happened to have a greater density of deciduous trees (Shirley 2004, 2005, 2006) which may partly explain why they often support more nesting bird species (Lock and Naiman 1998, Hagar 1999, Shirley 2004). Deciduous leaves often have higher nutrient levels (Roberts and Bilby 2009) and can support greater abundance and functional diversity of invertebrates (Piccolo and Wipfli 2002, Allan et al. 2003).

Studies are rare that compare buffer use by small mammals with small mammal use of undisturbed forests and clearcuts. Buffer studies involving larger mammals are even rarer. Changes in small mammal communities are notoriously difficult to separate from interannual variation due to climate (e.g., Hannon et al. 2002). A study in British Columbia found dusky shrews were less common in buffers than in larger forested areas. Significantly more deer mice and creeping voles were infested with bot flies at clearcut sites than at buffer sites, and none were infested at any of the control sites (Cockle and Richardson 2003). A meta-analysis of 69 small mammal buffer studies from throughout North America (Marczak et al. 2010) indicated that buffers tend to have lower densities of small mammals compared to larger undisturbed forests, but the results were not statistically significant and involved no species that is wetland-dependent in the Pacific Northwest.

3.3.8 Summary and Data Gaps: Effects on Microclimate, Vegetation, and Animals

The aforegoing review emphasizes that few if any peer-reviewed studies have measured directly the effects of various forest practices in the PNW on wetland microclimate, use by wetland-obligate birds and amphibian species, and viability of sensitive plants. Specific data gaps are described by the hypotheses in part 4 of Appendix A.

Tentative inferences have been made from the dozens of studies that have investigated logging effects on these ecosystem components in stream riparian areas of the PNW (e.g., Herlihey et al. 2005, Richardson 2008), as well as a few studies of timber harvests in or near wetlands in other northern regions (e.g., Palik and Kastendick 2010). What is most apparent from these or other studies is that removal of a significant part of a forest canopy in or around forested wetlands likely:

- increases light penetration and expands growth of algae in understory surface water, and consequently increases temporarily the abundance of aquatic invertebrates;

- heightens the risk of windthrow of trees;
- encourages invasion by non-native plants;
- eliminates some shade-intolerant plants and particular microclimate-sensitive mosses and lichens;
- facilitates colonization by wildlife species not present in the same forested wetland prior to harvest.

However, it is clear that no single, fixed-width buffer or canopy closure prescription will be adequate to protect all wetland-dependent species in all harvest operations (Richardson et al. 2012, Hruby 2013). Examination of buffer needs will have to consider scale-specific conditions and local landscape contexts.

4.0 Summary: Risks to Wetland Functions from Forest Practices

This review has clearly documented the paucity of well-designed (or any direct studies) of impacts to wetland functions from forest practices in the Pacific Northwest. This void is especially noticeable when placed in the context of the plethora of studies conducted of streams and riparian areas in this region. Despite the lack of direct studies of wetlands, this report has attempted to infer and hypothesize potential effects of forest practices on wetlands by considering what is known about forest practice effects on streams and, in a few cases, on different types of wetlands in other regions.

Perhaps the largest and **most probable** adverse effect of forestry operations on wetlands is the direct filling of small wetlands during the construction of logging roads. Although such roads occupy a seemingly small percentage of most watersheds, and road layout and designs have improved in recent years, the wetland losses that occur are essentially permanent. The extent of such losses has not been quantified at any scale in the PNW. At least historically, those losses were balanced somewhat by the acres of small wetlands that were created or expanded when logging roads intersected natural drainageways. A key question that remains is whether those created wetlands provide the same level of functions as natural wetlands.

With **reasonable certainty**, timber harvests in many PNW locations will result in a rise in local water tables, greater water yield, and warming of runoff. The degree to which these effects extend off-site and influence wetland functions depends on local geomorphic conditions. Also relatively probable is the fact that removal of a significant part of the forest canopy in or around forested wetlands will:

- increase light penetration and expand growth of algae in understory surface water, and consequently increases temporarily the abundance of aquatic invertebrates;
- heighten the risk of windthrow of trees;
- encourage invasion by non-native plants;
- eliminate some shade-intolerant plants and microclimate-sensitive mosses and lichens;
- facilitate colonization by wildlife species not present in the same forested wetland prior to harvest;
- increase soil erosion and export of suspended sediment from the logged site.

These effects usually diminish, in some cases exponentially, during the post-harvest years. At many locations the effects are undetectable after about 10 years post-harvest as vegetation succession and canopy closure occurs on logged sites. When canopy removal occurs outside a wetland boundary, the risk to a wetland depends mainly on the extent of canopy removal, its proximity to the wetland, and wetland size.

Most potential effects of forest practices on wetland functions have a moderate to low probability of occurrance. Rather than list all those here, they are described throughout this document and especially in the Hypothesis Appendix (A). Most notable among this category of potential effects are:

- the degree to which disturbances from logging operations mimic natural disturbances to wetlands in the PNW, and either add to or partially offset the effects of those on wetlands;
- effects of both wetlands and forest practices on the timing and seasonal persistence of surface water in downslope areas;
- effects of higher post-logging water tables on water detention times in wetlands, as that influences the pollutant processing capacities of wetlands, the reproductive success of pond-breeding amphibians, and fish access and growth;
- extent of small wetlands that experience faster sedimentation as a result of nearby logging operations, and the ecological consequences of that sedimentation;
- direct and indirect consequences (for salmon and other aquatic life) of changes (both immediate and as a progressive result of shorter rotations at catchment scales) in the amount, form, and timing of nutrients, large wood, and organic matter received and exported by wetlands;
- capacity of wetlands to continue to effectively process pollutants from logging runoff over the long term without adverse biological impacts to the wetlands;
- the optimal widths, configurations, and understory characteristics of upland buffers required for supporting particular wetland-dependent birds and mammals in various subregions and landscape settings;

- effects of logging-associated soil compaction, as well as canopy removal, on nonbreeding terrestrial habitat of wetland-breeding amphibians;
- the sublethal and indirect effects, in combination with other stressors, of commonly-used forest pesticides, on wetland fish, invertebrates, and amphibians;
- time required for tree stands to re-establish and regrow to their original condition after harvest in forested wetlands, and effects on other wetland functions attributable to possible time lags in this recovery;
- how all of the above are influenced by wetland type, connectivity, wetland position in a watershed, and other factors, especially factors that are subject to management;
- shifts that have occurred in gross vegetation structures (e.g., deciduous vs. conifer vs. herbaceous) across large portions of the region or watersheds, among forested wetlands that are recovering from logging many decades ago;
- shifts that have occurred in the mean size, connectivity, and distribution of wetlands of various hydrogeomorphic tyes or water regimes (e.g., proportionately fewer temporary-flooded vs. permanently-flooded wetlands; fewer lowland vs. headwater wetlands) across large portions of the region or watersheds, and the cumulative consequences of these shifts for wetland functions;
- the cumulative extent, compiled at multiple scales, of small wetlands in forested landscapes of the PNW, and the proportion of these wetlands exposed currently or historically to logging operations (on-site or off-site).

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