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Director's message

Anne C. Steinemann

I am delighted to write to you as the new Director of the Center for Water and Watershed Studies, and I am excited about our future together.

Since arriving, I have met scores of people who have told me what the Center has meant to them. Their testimonials are impressive.

Also impressive are the widespread commitments to the Center. For instance, this newsletter reaches more than 2,700 readers. Our Annual Review regularly attracts more than 300 attendees. We receive financial support from more than 20 local agencies.

The success of the Center can be measured by how we help our community and our constituency—for instance, how we address problems in the region, train students to be effective professionals, disseminate scientific information, and spark collaboration among researchers, practitioners, and the public.

The Center already has a strong regional reputation. My vision is for the Center to gain national recognition for its interdisciplinary and integrated research, education, and outreach on water and watershed issues. Our scientific studies help to inform decisions and solve problems. Real-world problems motivate research and educate students. The Center generates these synergies among people and organizations, matching issues with expertise.

In order to sustain and bolster the activities of the Center, I am actively pursuing funding sources that would provide a significant and consistent level of base support. In the funding pursuit, a key need is to demonstrate the Center's accomplishments and potential.

I need your input. Please write me (it's easy, see below) and tell me about two things:

(1) How has the Center has helped you, your organization, or your region? Please be as specific as possible—for instance, has the Center saved you money, provided resources, or generated other benefits? If possible, provide concrete examples of results, influence on decisions, or other outcomes.

(2) How can the Center help you in the future? Again, please be specific—can the Center address new problems, train students in certain areas, or provide more information on particular topics?

You may send e-mail to me at cwws@u.washington.edu. Please put "Input" in the subject line. Or, go to our webpage http://depts.washington.edu/cwws and click on the link for "Input."

Your responses will also be useful in the development of the Center's new strategic plan. More news on that will be coming soon.

The Center owes great thanks to Derek Booth, our outgoing Director, and Leslie Wall, our outgoing Program Coordinator. Derek and Leslie dedicated themselves to the Center, and we will carry forward and appreciate their countless contributions.

I have some good news: Leslie has agreed to stay with the Center, on a part-time basis, as we seek and train a new Program Coordinator. Leslie will also be developing a new website and public information program. Dan Ribeiro has agreed to come back to the Center and serve as our interim Program Coordinator and operations specialist. We are fortunate to have their experience and talent as we build the Center.

More good news: The Annual Review is "on" for this year—and it's free of charge to everyone. It will be held on February 17, 2005, at the same place—the HUB West Ballroom on the University of Washington campus.

Finally, I would like to thank you for your outpouring of support over the past months. I look forward to meeting you all and hearing from you. We will do great things together. ♦

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Incorporating uncertainty into freshwater habitat restoration

Jody Brauner Lando just finished her M.S. in Quantitative Ecology and Resource Management and Ph.D. in Aquatic and Fishery Sciences.

Recovery efforts for federally listed salmon stocks under the U.S. Endangered Species Act are widespread in the Pacific Northwest and highlight the difficulty of making decisions. As impacts to our natural resources continue to grow, the need to understand ecological systems and manage them effectively becomes increasingly important. For scientific knowledge to be useful to decision makers, information needs to be systematically linked in a framework useful to both scientists and policy makers. It is imperative that the analytical approach be transparent and address inherent uncertainties.

Uncertainty can stem from numerous sources including inaccurate or inadequate data, imperfect analytical methods, natural variation, and imprecise information transfer. When quantified and incorporated into biological analyses, uncertainty has been shown to change management decisions (Minns and Moore 2003). Fisheries science has embraced uncertainty analysis with extensive publications in the past 15 years; however, much of this work has focused on the advancing science of harvest management while overlooking the importance of uncertainty in freshwater habitat modeling. Therefore an analysis of epistemic uncertainty in riparian habitat modeling and its effects on management decisions was conducted. This article provides an overview of the methods used to conduct this study (Lando in press). While the data used are regionally derived (west of the Cascade range in Oregon and Washington), the concepts and techniques of this methodological study can be applied elsewhere.

Uncertainty

Uncertainty is present in our daily lives ranging from personal decisions (e.g., dressing for inclement weather, relationships, financial planning) to national and even global issues (e.g., economic development, the spread of HIV, threats of terrorism). Our awareness of these uncertainties enables us to develop safeguard strategies such as carrying an umbrella or diversifying an investment portfolio. Larger and more threatening uncertainties are not so easily solved, and frequently necessitate expert input from scientists and policy makers. Biological systems are no different. If we hope to conserve and effectively manage them, we must find ways to understand, incorporate, and respond to uncertainties.

Cullen and Frey (1999) formally defined uncertainty as “a measure of the incompleteness of one’s knowledge or information of an unknown quantity whose true value could be established if a perfect measuring device were available.” This should be expanded to consider an unlimited source of time and money necessary to establish a “true value.” By considering uncertainty in predictive models, decision-makers can consider the distribution of consequences, the impact of model assumptions, and the most productive direction for additional study. This study is based on four types of epistemic uncertainty commonly considered in fisheries science—process, observation, model, and estimation uncertainty.

Process uncertainty arises from natural variability. It incorporates the underlying stochasticity in population dynamics and the random variation in demographic rates and processes. Uncertainty in such models can obscure system dynamics, making correlative statistics impractical or inconclusive.

Observation uncertainty results from measurement and sampling error during the data collection process. Measurement error is the difference between the true value and our recorded observation, whereas sampling error stems from the probability that any selected sample is not completely representative of the population from which it is drawn. The incorporation of such types of error may obscure or create important fluctuations in the data.

Model structure uncertainty reflects our incomplete understanding and simplified representation of the processes and populations within a system (Fogarty et al. 1996). There is inevitable uncertainty in selecting an appropriate model to represent the dynamics of an ecological system and to predict future conditions. Rather than discovering the true model, we are simply finding one that fits the available data or reasonably predicts future conditions.

Parameter uncertainty arises when trying to determine the “true” value of a parameter in a mathematical model. The wide range of literature values and the complexity of ecological systems make this form of uncertainty prevalent in most physical models. Parameter uncertainty has been labeled a secondary type of uncertainty given that it can be generated from observation, process, and model uncertainty (Francis and Shotton 1997).

Ecological models

Quantifying uncertainty in riparian habitat management required the development and application of large woody debris (LWD) recruitment, pool formation, and coho habitat capacity models. Figure 1 shows the linkages between models.

LWD recruitment of Douglas-fir (*Pseudotsuga menziesii*) was modeled using Riparian Aquatic Interaction Simulator (RAIS)—a real-scale, deterministic model of wood recruitment linked to a forest growth and yield model (Welty et al. 2002). Developed for stands in western Oregon and Washington, RAIS predicts the growth of forest stands adjacent to streams and the subsequent amount and size of wood input to the stream channel over time. Six alternative management scenarios were considered including a range of thinning regimes along with an instream placement of LWD.

Models of pool habitat resulting from LWD recruitment were constructed based on previously published data collected in northwest Washington streams < 700 m in elevation and with channel slopes < 0.04 m/m (Montgomery et al. 1995; Beechie and Sibley 1997; Pess unpublished data). Surveys occurred during summer months and characterized stream width, gradient, LWD density, and habitat channel units according to Bisson et al. (1982). The resulting data set consisted of forty reaches from independent streams. Percent pool, a measure of wetted area classified as pool habitat per unit stream length, was selected as the index of measure and fit to each continuous predictor variable.

The final link between riparian habitat and stream function is expressed through predictions of juvenile coho rearing density. Insufficient winter habitat has been repeatedly documented to be a limiting factor in rearing juvenile coho (McMahon and Hartman 1989; Reeves et al. 1989; Solazzi et al. 2000), and thus served as the focus of this analysis. Juvenile coho densities by pool type and season were collected and published for a broad range of fully seeded streams along the Oregon coast (Nickelson et al. 1992). These data enable predictions of pool formation to be linked with coho density for the six riparian management scenarios. While the pool formation model generates physical capacity, the analysis assumed coho would fully utilize available territory and thus capacity estimates could be translated to density. Given the multitude of risks associated with an anadromous life history, the author acknowledges that populations are subject to other risk factors. Therefore "density" estimates are in fact "potential density" and should be treated as relative, rather than absolute values. The estimates assume other risk factors are held constant.

Analytical framework

The ecological models provide the foundation upon which to conduct the uncertainty analysis. Employing a decision analysis paradigm (Keeney 1977), there were seven distinct steps in the analytical framework

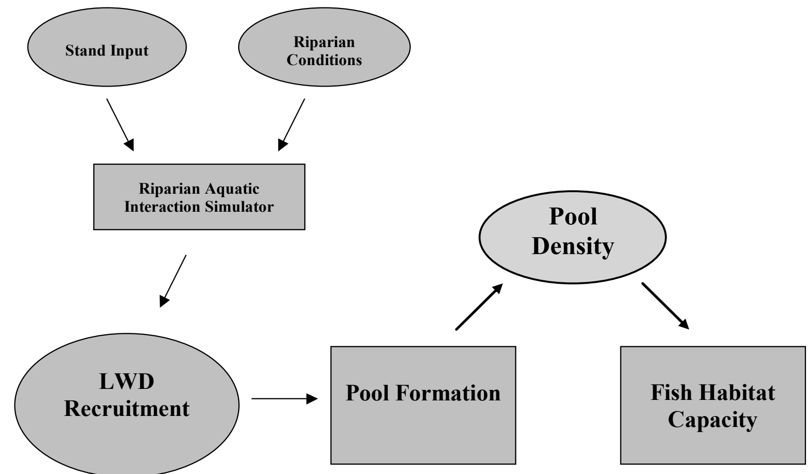

Figure 1. The linkages between physical and ecological models used to evaluate the impact of uncertainty in riparian restoration and salmon recovery. Circles reflect model inputs and outputs, while the rectangular boxes represent the actual models.

of this study. Collectively they provide a transparent process by which to evaluate the consequences of uncertainty given alternative management actions.

1. Identify riparian management scenarios.
2. Calculate the monetary cost per management scenario.
3. Determine likely sources of uncertainty.
4. Calculate the posterior probability of coho density given each source of uncertainty.
5. Define relevant decision makers.
6. Conduct a formal utility analysis.
7. Identify the optimal management scenarios.

The first step, identify riparian management scenarios, was constructed based on riparian harvest prescriptions and pool habitat conditions. Table 1 includes the riparian forest density, the assumed condition of pool habitat, and the cost for each scenario. Management scenarios 1, 2, and 3 included thinning practices and thus changed forest densities, whereas management scenarios 4, 5, and 6 held the stand density constant, but simulated restored stream conditions. The riparian forest densities served as input to the RAIS LWD recruitment model. The pool

Table 1. Management scenarios.

Management Scenarios	Thinning Regime (Trees per hectare)	Pool Habitat	Cost / km
1	73	Circa 1992	$ 7,078
2	57	Circa 1992	$ 5,413
3	30	Circa 1992	$ 2,082
4	73	5% restored	$ 63,528
5	73	15% restored	$ 186,528
6	73	30% restored	$ 371,028

habitat conditions were specified as a point of reference relative to the Nickelson et al. (1992) fish density data circa 1992. Lastly, the costs for each scenario reflect financial feasibility and were incorporated into a utility analysis for each decision maker.

Associated costs for riparian management were based on the stumpage value of trees removed during thinning (Bowers et al. 2002). Stumpage values were calculated on standing trees according to industry practice. Cost estimates for habitat restoration were based on a regional watershed planning document (Puget Sound Salmon Forum 2003). Large woody debris/engineered log jam placement was selected as the most suitable restoration strategy at a cost of $1,230 per lineal meter. This calculation assumed moderate stream size and LWD dimensions, as well as an average distance for site access.

Once the costs were calculated, the most likely sources of process, observation, model, and parameter uncertainty were considered. While there are certainly multiple sources for each type of uncertainty, only one was selected for this analysis. The dynamics of recruiting LWD were selected as the source of process uncertainty because of direct links to management practices. Sampling estimates of coho density were the selected form of observation uncertainty. This was motivated by recurrent literature documentation of sampling uncertainty in habitat surveys. While ecological model selection is commonly done on the basis of tradition or professional judgment, this study employed Akaike's Information Criteria (Akaike 1974) to objectively select candidate models and average the effects of model uncertainty. And lastly a Bayesian regression model with uniform prior parameters was constructed to estimate parameter uncertainty in pool habitat models. After considering the independent effects of each uncertainty, all four were collectively evaluated to establish a "compound" uncertainty.

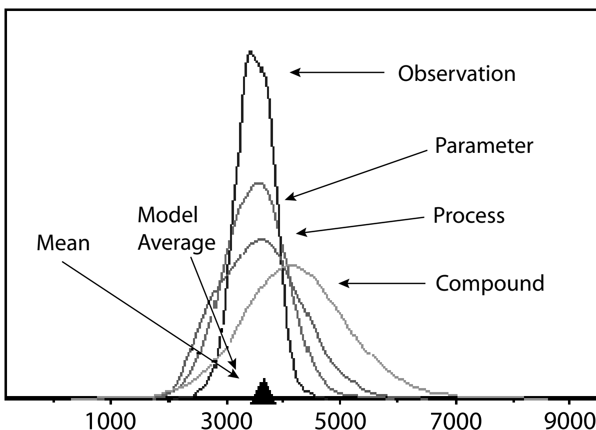

Figure 2. The rearing coho densities for all forms of uncertainty including the baseline of no uncertainty (mean) in year 50 for Manangement Scenario 5.

For the sake of a baseline comparison, multiple linear regression was used to estimate model parameters without uncertainty—a point estimate. Posterior probabilities of coho density were then generated using Bayes' theorem for process, observation, model, parameter, and a compound uncertainty. In all cases, prior probabilities were assumed to be uniform. As shown in Figure 2, compound uncertainty produced the widest range of rearing coho densities. In contrast, observation uncertainty produced the narrowest range of coho densities.

Rearing coho densities were then evaluated according to a utility analysis. Commonly used in statistics and economics, utility analysis provides an axiomatic framework of decision making under uncertainty. Utility is a metric of consumer preference expressed through the utility function—a desirability measure of management alternatives involving uncertainty (Keeney 1977). The analytical objective for a given suite of management alternatives is to maximize the utility function.

Based on historical involvement in recovery planning, six decision makers were identified and assigned utility functions (Figure 3A). While their preferences are hypothetically assigned, they reflect general trends in stakeholder participation. In order to construct the utility functions, a delisting criteria was established at 3000 juvenile coho/km stream reach. This target level affects public entities in different ways. For example, the Landowner cannot harvest any riparian stands until the density level reaches 2000 coho/km, therefore he has zero utility until that level is achieved. Likewise the Commercial and Recreational Fishermen have restrictions on their actions, which are reflected by asymptotically sigmoidal utility curves. A baseline preference is expressed by the Fish Advocate who consistently views more fish as better until the delisting criteria is met. And lastly the NOAA 1 and 2 decision makers reflect risk averse preferences. This convex form of a utility function shows diminishing marginal returns, which means an additional unit of coho density adds less to the total utility than the previous unit. This translates to higher values for increased coho rearing numbers at low densities and decreasing value as the densities approach delisting levels. Given the absence of current mandates, the delisting criteria were selected for demonstration purposes only.

In order to consider tradeoffs between alternatives, utility must also be assigned to the management costs. Thus a singular utility function for management costs was constructed to reflect an inversely proportional relationship between cost and utility (Figure 3B). Note the x-axis reflects decreasing management costs, standard protocol for structuring cost-based utility functions. It reflects a linearly inverse relationship between cost and utility.

In an effort to assign relative preferences between management cost and coho density, the following weights were applied for each decision maker.

NOAA 2 and the Fish Advocate placed 100% of their value on coho density, regardless of the costs incurred, whereas the Recreational Fisherman placed equal value on costs incurred and coho density. The other decision makers portrayed mixed weightings according to individual preference. The weights are used to generate a total utility reflecting both cost and coho density outcomes (Table 2).

In the same hypothetical construct as the utility functions of Figures 3A and 3B, the relative weighting assignments listed above were carefully constructed to reflect general trends in stakeholder preferences. For example, fisheries protection laws that govern NOAA representatives dictate value for species conservation, not management costs. That focused viewpoint resulted in the creation of NOAA 2 for whom 0% weight was placed on management costs. However, from a more realistic perspective, successful conservation must also be feasible, so the NOAA 1 decision maker portrays a 20% consideration for management costs. Another example is the equal weighting for the Recreational Fisherman. Logically the fisherman wants coho stocks to be restored in order to preserve his recreational interests, however he also needs to consider possible tax ramifications of supporting costly restoration scenarios. This conflicting perspective resulted in a 50% weighting for the utility of coho density and a 50% weighting for management costs. Along the same lines, the Commercial Fisherman also does not want to incur large personal tax increases from habitat restoration costs, but the impact of losing commercial fishing rights to a particular stock could be far more financially devastating, thus he allocates 90% weighting towards the utility of coho density.

Recognizing the potential effects of temporal variation, rearing coho densities for the point estimate and five sources of uncertainty were calculated at specific future time steps (20 and 50 years in the future). Given the fact that the optimal management scenario in year 20 may be different than the optimal scenario in year 50, the two time horizons were chosen to reflect the effects of short-term and long-term restoration perspectives. Next an expected total utility was calculated for each management scenario, time horizon, and decision maker. The management scenario with the largest total utility for a given decision maker is considered the optimal management choice. Table 3 shows that optimal choices for many stakeholders depending on the type of uncertainty and the time perspective. Results are compared relative to the optimal choices selected under no uncertainty (the first column). While some forms of uncertainty produced changes in management decisions, others did not. Compound uncertainty resulted in the greatest number of changes, followed by process and observation uncertainty. It is also noteworthy that most of the management changes selected more expensive alternatives.

Given the limited number of restoration options and decision makers (versus an infinite suite of possibilities) it is important to consider not simply the change in optimal management strategy, but also the change in expected utility. The value of considering expected utilities is discussed in detail in Lando (in press).

This analysis provides a logical and flexible framework within which to assign value to management alternatives. It is possible to change either the shape of a

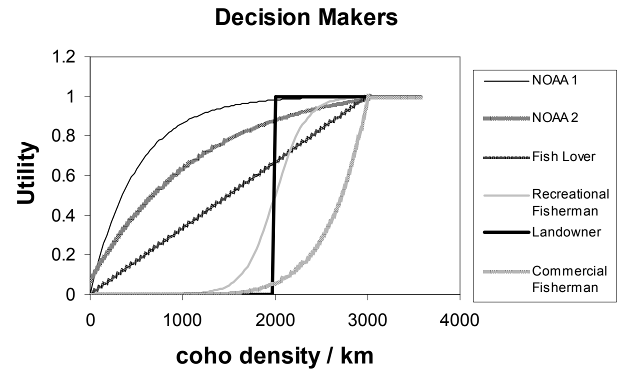

Figure 3A. Utility functions for coho density expressed according to specific decision makers.

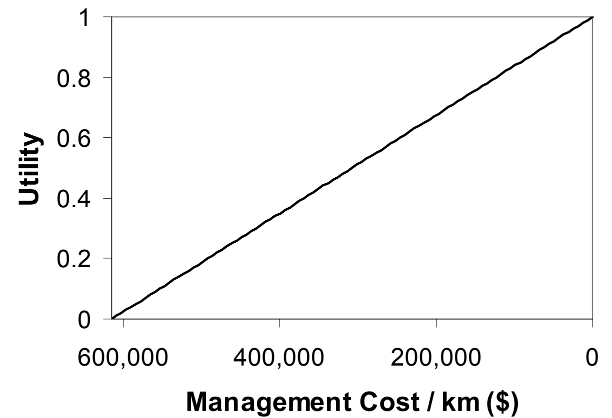

Figure 3B. Utility functions for management cost expressed according to specific decision makers.

Table 2. Utility weighting.

Decision Maker	Weight Management Cost	Weight Coho Density
NOAA 1	0.2	0.8
NOAA 2	0.0	1.0
Landowner	0.1	0.9
Commercial Fisherman	0.1	0.9
Recreational Fisherman	0.5	0.5
Fish Advocate	0.0	1.0

Table 3. Optimal choice. Each cell of the table identifies the optimal management scenario at year 20 (uppermost number) and year 50 (lowermost number) based on the expected total utility for a particular decision maker.

Decision Makers	Source of Uncertainty					
	None	Process	Observation	Model	Parameter	Compound
NOAA 1	1, 4 1, 4	1, 4 1	1, 4 1	1, 4 1, 4	1, 4 1, 4	4 1, 4
NOAA 2	6 1, 4, 5, 6	6 1, 4, 5, 6	6 1, 4, 5, 6	6 1, 4, 5, 6	6 1, 4, 5, 6	4, 5, 6 1, 4, 5, 6
Landowner	4 4	6 6	6 6	1 4	6 6	4 5, 6
Commercial Fisherman	6 4	6 6	6 5	6 4	6 6	6 5
Recreational Fisherman	5 1	5 5	5 1	5 1	5 1	4 1
Fish Advocate	6 4, 5, 6	6 6	6 5, 6	6 4, 5, 6	6 6	6 5, 6

particular utility function or the relative weighting assignments. Such an explicit and methodological analysis illustrates both the process by which to incorporate uncertainty and the management implications. It provides a powerful set of tools by which to increase biological realism and reflect stakeholder preferences.

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Snapshot of current research

Relative contributions of common native and non-native vegetation species to water nutrient concentrations

Mindy Roberts, Ph.D. student, Center for Water and Watershed Studies and Civil and Environmental Engineering

Take 500 mL of filtered stream water. Add 1 to 2 grams of dried leaf material. Stir. Wait. A regional recipe for tea?

To understand the fate of leaf material in small stream systems and the role of allochthonous materials in stream nutrient and energy regimes, I conducted a laboratory experiment to compare differences in the short-term nutrient dynamics among native and non-native species found along Puget Lowland streams. The experiment mimics the deposition of litter into streams and subsequent transport into microenvironments such as pools and debris jams where the organic matter decomposes.

Dried materials were added to beakers with filtered stream water, collected from a groundwater-dominated urban stream with high background nitrate levels, held at 10° C. Water and leaf samples were collected over a 72-hour period.

Nitrate decreased and ammonium increased in beakers containing moss, bigleaf maple, Himalayan blackberry, Japanese knotweed, and red alder (Figure 1A). Ammonium levels increased initially in beakers containing Western hemlock and red cedar but decreased to near background levels after 72 hours. English ivy and holly did not change ammonium concentrations significantly. Leaf nitrogen content did not explain the variation in water nitrogen levels. Organic matter with a tough waxy cuticle (English ivy, holly, red cedar, Western hemlock) appears to reduce nutrient exchange.

Orthophosphate concentrations in beakers with organic matter increased over time to levels greater than those found in beakers with source water only. The addition of red alder and Japanese knotweed leaves, which contain the highest levels of phosphorus of the species studied (1.3 and 1.7 mg/kg, respectively), produced the largest increases (Figure 1B). Like nitrogen, however, the initial leaf phosphorus content did not explain the variation in water orthophosphate levels, since Western hemlock and English ivy leaves contain more phosphorus (1.3 and 1.1 mg/kg, respectively) than bigleaf maple and Himalayan blackberry (0.9 and 0.5 mg/kg, respectively) but produced lower orthophosphate concentrations.

What does it all mean? Leaves could increase dissolved nutrient concentrations significantly, as indicated by comparing levels in beakers with and without organic matter, through leaching and as substrates for biological activity. Common native and non-native species contribute differently to water nutrient levels, and the relative effects are not explained by initial leaf nutrient levels. Native deciduous species (red alder and bigleaf maple) as well as two common non-native species (Japanese knotweed and Himalayan blackberry) produce greater ammonium and orthophosphate levels than dominant coniferous species red cedar and Western hemlock. While these results are preliminary and related studies continue, they suggest that as riparian vegetation changes from historically mixed forests to stands dominated by red alder or non-native species such as blackberry and knotweed, we may fundamentally change stream nutrient dynamics. ♦

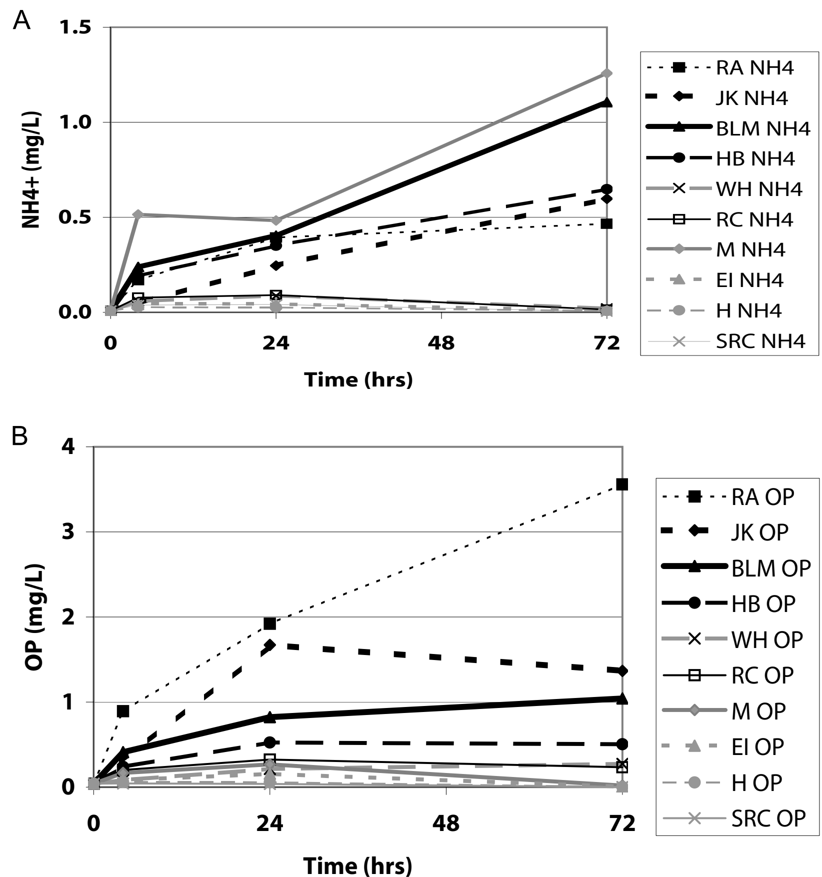

Figure 1. Ammonium and orthophosphate levels increase over time in vessels containing organic matter, with red alder (RA), Japanese knotweed (JK), bigleaf maple (BLM), and Himalayan blackberry (HB) producing the largest increases compared to source water (SRC). Western hemlock (WH) and red cedar (RC) produced smaller increases in concentrations, while moss species (M), English ivy (EI), and holly species (H) produced little change.

Thesis and dissertation abstracts

Below is an abstract from a recently completed thesis of an affiliated graduate student. The web site has a list of all affiliated students who have graduated, many of their abstracts, and some entire theses or dissertations (http://depts.washington.edu/cwws/Theses/thesesabstracts.html).

An assessment of large woody debris and riparian forest resources at Ellsworth Creek Watershed and a comparison of riparian management options

Steven Rentmeester, M.S., Aquatic and Fishery Sciences

Selecting and prioritizing large woody debris enhancement projects requires the ability to stratify instream habitat and riparian forests based on current conditions and ecosystem recovery pathways. During the summer of 2003, we inventoried large woody debris, habitat, and geomorphological characteristic of all fish bearing water at Ellsworth Creek Watershed (24.1 km). Additionally, the spatial distribution, stand structure, and species composition of riparian forests along main-stem channels were characterized using aerial photos and field data. Trees, saplings, and topography were measured within fifty-seven, 500 m^2 riparian forest transects. Headwater and main-stem channels (15.9 km and 8.2 km, respectively) were delineated along a 200 ha upstream drainage area breakpoint. As compared to unmanaged basins, headwater channels at Ellsworth Creek had higher than expected abundance of instream wood and key pieces (p-values < 0.001). Main-stem channels were deficient in wood abundance, volume, and key pieces (p-values < 0.001).

Riparian forests within 50 meters of main-stem channels occupy 77.8 hectares. The history of road building and timber harvest left a distinct spatial pattern on these forests. Stands younger than 20 years old accounted for 19.1% of the total area. Stands 20–40 years old covered 22.4%, 40–80 year-old stands covered 33.3%, and stands older than 200 years covered 23.2% of the total area. Average basal area ranged from 24.6 m^2/ha in 15 year-old stands to 70.2 m^2/ha in stands greater than 200 years old. Stem density ranged from 700–2480 TPH in 25 year-old and younger stands. Density ranged from 140–1000 TPH in older stands. Quadratic mean diameter increased from 14.3 cm in 15 year-old stands to 43.2 cm in stands greater than 200 years old. Channel confinement and stream adjacent landform were important factors influencing species composition.

Effects of thinning on riparian forests were simulated using the Pacific Northwest Coast variant of the Forest Vegetation Simulator (FVS-PNC). Thinning from below increased production of large diameter snags by 28–74% over a "no touch" silviculture. Although thinning from below increased diameter growth in the simulations, it has also been shown to increase wind firmness and has the potential to decrease instream wood recruitment originating from wind disturbance. ♦

Upcoming events

Details for these events can be found at http://depts.washington.edu/cwws/Outreach/Events/seminars.html

January 4 – March 8, 2005	Tuesday Morning Seminar Series, 8:30 to 9:30 am, 22 Anderson Hall, UW Campus
January 6 – March 17, 2005	Monster Seminar Jams, co-sponsored with NOAA Northwest Fisheries Science Center, 11 am to 12 pm, 2725 Montlake Blvd E.
February 17, 2005	Annual Review of Research, HUB West Ballroom, UW Campus