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LINKING LANDSCAPE CHARACTERISTICS AND HIGH STREAM NITROGEN IN THE OREGON COAST RANGE: RED ALDER COMPLICATES USE OF NUTRIENT CRITERIA¹

Effie A. Greathouse, Jana E. Compton, and John Van Sickle²

ABSTRACT: Red alder (*Alnus rubra*), a nitrogen(N)-fixing deciduous broadleaf tree, can strongly influence N concentrations in western Oregon and Washington. We compiled a database of stream N and GIS-derived landscape characteristics in order to examine geographic variation in N across the Oregon Coast Range. Basal area of alder, expressed as a percent of watershed area, accounted for 37% and 38% of the variation in summer nitrate and total N (TN) concentrations, respectively. Relationships between alder and nitrate were strongest in winter when streamflow and landscape connections are highest. Distance to the coast and latitude, potential surrogates for sea salt inputs, and watershed area were also related to nitrate concentrations in an all-subsets regression analysis, which accounted for 46% of the variation in summer nitrate concentrations. The model with the lowest Akaike's Information Criterion did not include developed or agricultural land cover, probably because few watersheds in our database had substantial levels of these land cover classes. Our results provide evidence, at a regional scale, that background sources and processes cause many Coast Range streams to exceed proposed nutrient criteria, and that the prevalence of a single tree species (N-fixing red alder) exerts a dominant control over stream N concentrations across this region.

(KEY TERMS: Pacific Northwest; biogeochemistry; nutrients; rivers/streams; forests; environmental regulations.)

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INTRODUCTION

Human-mediated changes in nutrient cycles degrade a variety of ecosystem goods and services valued by people (Schlesinger, 1997). For example, eutrophication of aquatic ecosystems causes harmful algal blooms, fish kills, and poor quality of sources of

drinking water. Addressing these problems requires empirical research that relates nutrient chemistry to landscape characteristics both correlatively and mechanistically (Gergel *et al.*, 2002). For landscape-nutrient relationships in western United States (U.S.) streams, significant and predictive empirical relationships and refinement of nutrient loading models are likely to depend on different and/or more

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detailed land cover classifications than the coarse land cover categories that have often shown high correlation with chemistry of eastern streams (Jones *et al.*, 2000). Tree species composition may play especially important roles in controlling stream nutrient levels in particular regions of the west, similar to findings in subregions of the east, such as the Catskill Mountains in New York State (Lovett *et al.*, 2002). Predictions that western streams will show reduced nutrient levels with increasing percent forest land cover, similar to many eastern streams (Jones *et al.*, 2000), are likely inaccurate for parts of the Pacific Northwest (PNW) because these types of coarse land cover categories do not account for the widespread and spatially variable occurrence of red alder (*Alnus rubra*) (Wigington *et al.*, 1998; Binkley *et al.*, 2004), an early successional nitrogen(N)-fixing deciduous broadleaf tree commonly found following disturbance (Long *et al.*, 1998; Long and Whitlock, 2002).

N fixation by red alder, which occurs in root nodules containing symbiotic actinomycetes (*Frankia* spp., Hibbs *et al.*, 1994), is particularly important to consider in the Oregon Coast Range (OCR). Oregon's coastal mountains span an area of ~52,000 km², and red alder is the second most abundant tree species in the region (Ohmann and Gregory, 2002). Estimates of fixation rates by red alder are remarkably high in the OCR (e.g., generally 100-200 kg N/ha/yr in pure stands and 50-100 kg N/ha/yr in mixed conifer-alder stands), and N is fixed to the point where it accumulates and leaks out of alder stands at high rates (Hibbs *et al.*, 1994). Soils of the OCR have very high N, occupying the top 5% of soil N content worldwide, and the legacy of cycles of fire and red alder succession appear to drive these high soil N levels (Perakis *et al.*, 2011). Likewise, a history of alder cover in OCR conifer forests can also lead to high N leaching (Perakis and Sinkhorn, 2011).

Previous research suggests but leaves open the question of whether an effect of alder on stream N levels actually occurs across the OCR. Wigington *et al.* (1998), in a study of chemistry in 48 OCR streams, hypothesized that alder was the primary control of spatial variation in nitrate concentrations but found little correlation with the limited indicators of alder available at the time. They found no correlation with percent hardwood cover and only moderate correlation with percent hardwood-conifer cover determined from satellite data. Using more refined landscape data, but looking at a scale much smaller than the OCR (subbasins of the OCR Salmon River watershed), Compton *et al.* (2003) showed that hardwood cover was dominated by alder and explained a high proportion of the variance in N concentrations and N export in this watershed. A

number of other studies at plot to watershed scales have also indicated that alder species increase N levels in aquatic systems (e.g., Stottlemeyer and Toczydlowski, 1999; O'Keefe and Edwards, 2002; Sigleo *et al.*, 2010; for a comprehensive list of studies by geographic region, see Table 1 in Appendix S1), but evidence for alder effects on stream N is only moderate in many of these studies (e.g., comparisons of a single high-alder watershed to a single low-alder watershed), and none were conducted across a broad regional scale.

To put previous alder-aquatic N studies that were spatially limited into a broader regional context, we used data from various monitoring and GIS efforts to compile a database of stream N concentrations and landscape characteristics in the OCR. Compiling this spatially extensive and representative database, and examining correlations with GIS layers, allowed us to: (1) test the hypothesis that OCR stream nitrate concentrations increased with watershed alder cover and (2) to evaluate whether these alder-specific data generated by a novel GIS technique (the gradient nearest neighbor, or GNN, method developed by Ohmann and Gregory, 2002) were better than coarse geospatial vegetation data from the National Land Cover Dataset (NLCD) at explaining stream N. We also compared N levels in forested streams to proposed nutrient criteria and used multiple regressions to examine seasonal patterns and explore relationships with landscape variables other than alder, including possible indicators of sea salt inputs and urban and agricultural impact.

METHODS

Data

Compilation and Treatment of Stream N Data. We obtained stream N data for 761 sites sampled from 1990 to 2007 in the Oregon portion of the Coast Range ecoregion (Level III ecoregion, Omernik, 1987; Figure 1); 593 of these sites were both freshwater streams (i.e., were not estuarine- or beach/dune-influenced sites) draining watersheds within the OCR and had data on analytes (nitrate/nitrite-N, total N [TN], and total Kjeldahl N [TKN]) with adequate spatial representation and detection limits (Table 1 and Appendix S1). To confirm comparability of analytes across studies, we obtained documentation of field procedures, as well as QA/QC and standard operating procedures from the three analytical laboratories which conducted stream N analyses (references in Table 1 and Appendix S1). All three laboratories

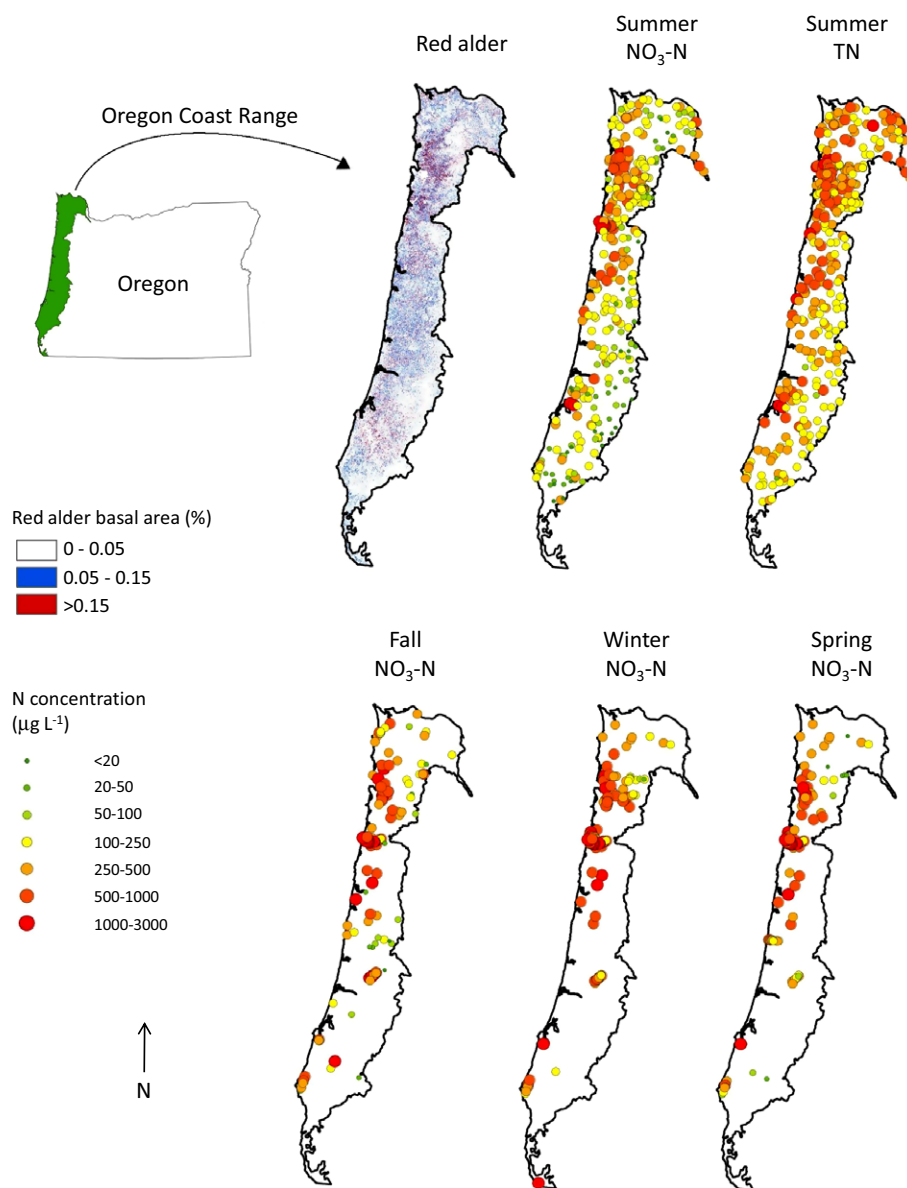


FIGURE 1. Basal Area of Red Alder and Nitrogen Data by Season across the Oregon Coast Range. Seasons are calendar-based summer, fall, winter, and spring. The legend for N concentrations applies to both $\text{NO}_3\text{-N}$ and TN data displayed in the maps.

analyzed nitrate/nitrite-N by cadmium reduction, and we refer to nitrate/nitrite-N as $\text{NO}_3\text{-N}$ in this study. Unfiltered TN was analyzed by persulfate digestion at the Willamette Research Station and the Cooperative Chemical Analytical Laboratory (CCAL, at Oregon State University, OSU). Unfiltered TKN was analyzed by automated phenate at the Oregon Department of Environmental Quality Laboratory (OR DEQ Lab). Analytes were converted to standard units ($\mu\text{g N/l}$). Looking at overlapping time series of data collected by different laboratories at a single site in the Salmon River basin further confirmed that nitrate data collected by different laboratories were comparable.

We grouped sites with direct measurements of TN (i.e., persulfate digestions described above) together with sites at which TKN and $\text{NO}_3\text{-N}$ measured in the same sample could be added to obtain an estimate of TN (cf., Patton and Kryskalla, 2003; and see Appendix S1). For $\text{NO}_3\text{-N}$ data, detection limits varied from 2 to 20 $\mu\text{g N/l}$. The detection limit for TKN data was 200 $\mu\text{g N/l}$. No TN data were below detection. For below detection $\text{NO}_3\text{-N}$ and TKN values, we used machine-read values, if available, but if machine-read values were not available, we substituted half the detection limit (cf., Antweiler and Taylor, 2008). When multiple samples were taken at a single site in a single year or multiple years, we averaged those

TABLE 1. Projects with Stream Nitrogen Data in the Oregon Coast Range (1990-2007) — Data Collated for This Study from Freshwater Streams Draining Watersheds within the Oregon Coast Range.

Project Name, Primary Agency/Organization ¹	Number of Sites by Analyte			Dates	Data Source/Selected Citations ²
	NO ₃ -N	TKN	TN		
USEPA WED projects, WRS ³					
Freshwater Habitat Project	88			2000-2006	Compton <i>et al.</i> (2003); Ebersole <i>et al.</i> (2009)
EMAP projects					Herlihy and Sifneos (2008)
Oregon Streams and Rivers 1997	23		23	1997	Lazorchak <i>et al.</i> (1998)
Oregon Rivers 1998	3		3	1998	Lazorchak <i>et al.</i> (2000)
EMAP-West	10		10	2000-2003	Stoddard <i>et al.</i> (2005)
Oregon Tillamook Kilchis 1998-1999, OSU			52	1998-1999	Naymik <i>et al.</i> (2005)
OR DEQ projects, OR DEQ Lab ³					
REMAP Coast Range	42	42		1994-1996	Herger and Hayslip (2000)
Ambient River Monitoring	14	14		1990-2007	Cude (2001); Mrazik (2006)
Reference Site Monitoring	45	45		1990-2007	Drake (2004)
Oregon Salmon Plan	138	136		1998-2007	OR DEQ (2005)
Other project/general sampling	164	154		1990-2007	OR DEQ (2005)
Other organizations, CCAL ³					
Trask Watershed Study, OSU	29			2006	Ashkenas and Johnson (2007)
AREMP, USFS		6		2001-2003	Gallo <i>et al.</i> (2005)

¹Agencies/organizations not defined in text: WED, Western Ecology Division; AREMP, Aquatic & Riparian Effectiveness Monitoring Program; USFS, United States Forest Service.

²Data source/selected citations lists citations and researchers we obtained data from. Additional details on databases used and additional studies with OCR stream nitrogen data not used, obtained or collated in our study are available in Appendix S1.

³Analytical laboratory procedures: WRS: Motter and Gruen (2007a, b). OR DEQ Lab and projects sampling references: OR DEQ (1998, 2002a, b, 2004a, b, 2006). CCAL: Motter and Jones (2006a, b).

data such that we ended up with a single seasonal value for each site (based on calendar season).

GIS Layers and Predictor Variables. We chose landscape variables based on likely importance to stream N as well as practicality (i.e., what we were able to determine from existing GIS layers). The majority of our watershed delineations were obtained from the OR DEQ or an internal U.S. Environmental Protection Agency (USEPA) database of Environmental Monitoring and Assessment (EMAP) GIS data. For sites that were not delineated by EMAP or OR DEQ, watersheds were delineated using the same methods that OR DEQ and the EMAP program used: coordinates obtained from original data sources (see Table 1 and Appendix S1), applied to 10-m digital elevation models from the Coastal Landscape Analysis and Modeling Study (CLAMS, 2006) or from the Oregon Bureau of Land Management for areas not covered by CLAMS (Oregon Bureau of Land Management, 1998).

A novel alder-specific metric, as well as other vegetation and land cover variables, were determined from a map produced by the LEMMA group (Landscape Ecology, Modeling, Mapping and Analysis, 2008; 2000 GNN/IMAP Coast Range vegetation layer). The primary vegetation data provided by this 2000 GNN/IMAP layer are forest species composition

and basal area (i.e., the area of the cross section of tree trunks at breast height, for each tree species occurring in each 30 × 30 m cell of the map). The LEMMA group produced this vegetation data using the gradient nearest-neighbor mapping approach of Ohmann and Gregory (2002), whereas land cover data from the GNN/IMAP layer are based on a modification of the NLCD (Grossman *et al.*, 2008). Watershed delineations were used to clip this coverage, to estimate the following for each watershed: total basal area of alder (m²), total basal area of other hardwoods (m²), developed land cover (ha), agricultural land cover (ha), and natural land cover (defined below, ha). These variables were then expressed as percents of watershed area to calculate both the alder metric (basal area of alder as a percent of watershed area: Figure 1 and Table 2) and other GNN-based vegetation and land cover metrics (basal area of other hardwoods as a percent of watershed area, watershed-level percent natural land cover, watershed-level percent developed land cover, and watershed-level percent agricultural land cover: Table 2). Developed land cover was the sum of four developed classes ranging from open space to high intensity. Agricultural land cover was the sum of two classes (pasture/hay and cultivated crops). Natural land cover represented all of the forest classes (Vegclass codes 1-11) plus all of the natural nonforested land cover categories (see Appendix S1).

TABLE 2. Statistics for Sites with Nitrate and Total Nitrogen (TN) Data in Summer.

	NO ₃ -N (<i>n</i> = 386)					TN (<i>n</i> = 336)				
	Mean	Median	Min	Max	<i>r</i>	Mean	Median	Min	Max	<i>r</i>
Basal area of alder (%) ¹	0.036	0.032	0.000	0.140	0.61	0.036	0.032	0.000	0.105	0.60
Latitude	44.73	45.03	42.57	46.15	0.37	44.7	45.18	42.57	46.15	0.32
Distance to coast (km)	26.52	25.50	0.16	92.40	-0.33	26.73	23.04	0.16	92.40	-0.31
Basal area of other hardwoods (%) ¹	0.016	0.010	0.000	0.165	-0.33	0.016	0.011	0.000	0.165	-0.26
Elevation (m above sea level)	178.3	132.8	1.7	824.9	-0.32	163.0	131.5	3.1	824.9	-0.44
Watershed-level developed land cover (%) ¹	5.3	4.9	0.0	31.5	0.12	5.3	4.9	0.0	31.5	0.20
Watershed-level natural land cover (%) ¹	94.2	94.7	68.5	100.0	-0.12	94.2	94.7	68.5	100.0	-0.26
Watershed area (ha)	5,028	818	9	185,325	-0.11	5,331	1,081	9	185,325	-0.12
No. of CAFOs ²	0.1	0	0	13	0.07	0.2	0	0	13	0.16
Watershed-level agricultural land cover (%) ¹	0.4	0.2	0.0	15.4	0.05	0.5	0.02	0.0	15.4	0.21
Site-level land cover type ³	—	—	—	—	-0.04	—	—	—	—	-0.13
No. of lakes and reservoirs ²	0.1	0	0	6	<0.01	0.1	0	0	6	-0.02
Average summer value (µg N/l)	266.4	186.2	0.0	2,235.7	—	401.7	323.2	33.0	2,075.0	—

Notes: Table gives means, medians, minimums (min), maximums (max), and Pearson product-moment correlations coefficients (*r*) between N variables and landscape variables. Percentage data were arcsine-square root-transformed and nitrate, TN and watershed area were log transformed before figuring *r*.

¹As a percent of watershed area.

²Mean, median, min, and max values are as absolute numbers per watershed; correlation coefficient was calculated against numbers per unit watershed area; CAFO, confined animal feeding operation.

³For the NO₃-N dataset, site-level land cover type at 348 sites was judged to be natural, 28 had agriculture, 9 had urban land cover, and 1 site had agriculture and urban land cover; for TN, 297 sites were natural, 31 had agriculture, and 8 had urban land cover.

We also used clips of the 2001 NLCD (Homer *et al.*, 2004) to obtain another set of vegetation metrics that researchers may attempt to use as surrogates of alder cover: the coarse vegetation data we obtained from the NLCD map were percent mixed forest and percent deciduous forest. Stream N data spanned a 17-year period; however, because we lacked temporal resolution in the landscape data, we used the 2000 GNN/IMAP and 2001 NLCD layers as representative snapshots of vegetation cover and land use/land cover. Alder stands persist for 80-120 years (Hibbs *et al.*, 1994), suggesting that a snapshot in the middle of this 17-year period is an adequate measure of alder cover for the period.

We characterized the alder metric in the whole watershed only because Compton *et al.* (2003) found that stream chemistry in the Salmon River basin (OCR) was more strongly related to hardwood cover in the entire subbasin than to hardwood cover in the riparian zone alone. Detailed comparisons of site level *vs.* whole watershed land use levels were likewise outside the scope of our analysis. However, to avoid missing possible signals of agricultural or urban land cover impacts on stream N, we examined both whole watershed land cover and land cover at the site because no prior research indicates whether OCR stream chemistry is more affected by local scale *vs.* whole watershed levels of developed and agricultural

land uses. Moreover, because agriculture and development are concentrated in lowland valleys in the OCR, sites could be located, for example, in the middle of a dairy farm but have a high level of forested land cover in the watershed. Thus, in addition to the whole watershed land cover classifications (i.e., alder-specific metric, other GNN-based vegetation and land cover metrics, and coarse vegetation data from the NLCD), we characterized local site land cover. Site-level land cover type was characterized visually by the lead author. Aerial photos were viewed at a scale of 1:16,000, and land cover in an approximate 200-m radius half circle upstream from the site was categorized as natural, urban, agricultural, or a combination of these three categories (i.e., if the entire half circle upstream from the site appeared to be vegetated, the site was categorized as having natural site-level land cover; if the half circle appeared to contain any urban or agricultural land use, the site-level land cover was categorized as urban or agricultural or both urban and agricultural).

Additional site and watershed geospatial characterization was undertaken for distance to coast, number of CAFOs (confined animal feeding operations), and number of lakes and reservoirs in the watershed. Distance to coast was a straight line distance between the site coordinates and a coastline based on a layer of Oregon counties with estuaries removed.

Number of CAFOs in the watershed was determined from a 2007 version of the Oregon Department of Agriculture's GIS layer (see Appendix S1). Number of lakes and reservoirs in the watershed were determined using the NHDLakesPonds and NHDReservoirs layers from the USGS National Hydrography Dataset (U.S. Geological Survey, 2012).

Data Analysis

Overall Approach. Our primary interest was to test the hypothesis that stream N concentrations increase with alder across the 52,000-km² OCR, similar to findings in our laboratory's previous research showing stream N increased with alder in the Salmon River watershed, a minimally disturbed 200-km² watershed within the northern part of the OCR (Compton *et al.*, 2003). However, we were also interested in other questions and patterns in the collated data, including comparisons of N data to proposed nutrient criteria, what alder-N patterns look like if we use different possible indicators of alder cover (e.g., the GNN alder metric *vs.* coarse vegetation data from the NLCD), seasonal patterns in N-alder relationships, and relationships between N and variables other than alder. Thus, before conducting any data analyses, we decided on one initial *a priori* set of hypothesis tests for examining our primary question of whether stream N increases with alder across the OCR. We also decided to limit our use of formal hypothesis testing using ANOVAs and *p*-values in other analyses to avoid an overreliance on reporting *p*-values without formulating strong *a priori* hypotheses (cf., Burnham and Anderson, 2002); our only use of *p*-values in other analyses was in our analysis of seasonal data.

Thus, for our overall approach, we conducted three main sets of analyses, described in more detail in sections below, all three of which had a primary focus of relating alder to stream NO₃-N. First, we looked at alder influences on summer N. In this first set of analyses (described in detail in the section below "Alder Influences on Summer N Concentrations"), we conducted our formal set of hypothesis tests using summer NO₃-N and the GNN alder metric (basal area of alder as a percent of watershed area), with data subsetted by watershed-level percent natural land cover, site-level land cover type, and number of CAFOs in the watershed in order to identify minimally disturbed sites. We also regressed summer TN against the alder metric in comparison to nutrient criteria, examined distance to the coast as a potential confounding factor, and related potential surrogates of alder (coarse vegetation data from the NLCD) to summer NO₃-N. Second, we conducted a seasonal

analysis, relating NO₃-N data to the GNN alder metric across four seasons (see section below "Seasonal Analysis of NO₃-N"). Third, we conducted exploratory analyses of summer N, looking at relationships with variables in addition to the alder metric (see section below "Exploratory Analyses of Summer N"); the additional variables examined in exploratory analyses were the potential surrogates of sea salt inputs (distance to the coast, latitude), the other GNN-based vegetation metrics (basal area of other hardwoods as a percent of watershed area, watershed-level percent natural land cover), the land cover/land use metrics (watershed-level percent developed land cover, watershed-level percent agricultural land cover, site-level land cover type, number of CAFOs), and other watershed characteristics (number of lakes and reservoirs in the watershed, watershed area, elevation).

Before conducting analyses, we assessed spatial independence at our sites by plotting between-site differences in NO₃-N (log *x* + 1-transformed, see below for information on transformation) against Euclidean (straight-line) between-site distances, for all possible pairs of sites. The resulting scatterplot (a variogram cloud, Bivand *et al.*, 2008) showed no evidence of smaller NO₃-N distances at smaller Euclidean distances, thus indicating spatial independence of our data. However, nested watersheds in the dataset still presented a conceptual basis for the dataset containing sites which were not independent. Thus, we removed nested sites that were within 5 km of each other (cf., Johnson *et al.*, 1997) and had watershed areas that were less than twice as large as the smallest of the nested watersheds. After removing these nested sites, we had 482 sites with NO₃-N data (any season) and 336 sites with summer TN data (270 estimated by adding TKN and NO₃-N; 66 measured directly by persulfate digestion) that we considered independent and suitable for analyses. We conducted most of our analyses on summer data (as defined by the calendar season: NO₃-N = 386 sites; TN = 336 sites) because fewer data were available in other seasons (Figure 1) and because nutrients have the most potential for negative impacts when stream ecosystems have their highest temperatures and lowest flows (Allan, 1995). We also conducted most of our analyses on NO₃-N data because the TN data presented complexities (e.g., multiple types of TN data with estimates from TKN and NO₃-N and measurements by persulfate digestion) and weakness (e.g., high detection limits).

Statistical analyses described below were conducted in R (version 2.11.1) and SAS (version 9.2, Cary, North Carolina). To make residuals normal with constant variance, and meet assumptions of linear relationships for regressions, we used log(*x* + 1)-transformed NO₃-N and TN (noted in equations as logNO₃-N and logTN,

respectively), log-transformed watershed area, and arcsine-square root-transformed percentage data (with alder as a percent of watershed area noted as *alder* in equations) (see Pan *et al.*, 2004 and Stanley and Maxted, 2008 for similar use of these transformations with N- and GIS-derived percentage data). Regression diagnostic plots (e.g., residual plots, *Q-Q* plots, residuals *vs.* leverage plots which included Cook's distance), plots of histograms, and plots of the transformed data (not shown) indicated that transformations successfully normalized data, made variance constant, and resulted in linear relationships.

Alder Influences on Summer N Concentrations. Before conducting any other statistical analyses, we used linear regression to test our main hypothesis that alder levels are associated with stream $\text{NO}_3\text{-N}$ in minimally disturbed watersheds in the OCR. This hypothesis was based on findings of Compton *et al.* (2003) for the Salmon River basin, a small OCR basin where stream sites had little human influence and where hardwood cover as a surrogate of alder was previously shown to correlate with stream $\text{NO}_3\text{-N}$. Thus, to examine this hypothesis across the entire OCR and to ensure that human influences did not obscure a relationship between the GNN alder metric and $\text{NO}_3\text{-N}$ in this hypothesis test, we used the subset of summer $\text{NO}_3\text{-N}$ sites that had the following characteristics of sites in the Salmon River basin study: watershed-level percent natural land cover was greater than 90%, site-level land cover type was natural, and there were no CAFOs present in the watershed (see Rohm *et al.*, 2002; Smith *et al.*, 2003; Herlihy and Sifneos, 2008, for other examples of subsetting data by characteristics of human influence to examine patterns in minimally disturbed watersheds).

We also looked at the relationship between alder and $\text{NO}_3\text{-N}$ without any criteria for natural land cover and CAFOs, in order to examine the relationship when we included summer $\text{NO}_3\text{-N}$ data from all 386 independent sites. Because results for $\text{NO}_3\text{-N}$ in both the hypothesis test and this examination of all 386 independent sites were extremely similar (see Results section) and because in our subsequent analyses we did not have the specific goal of examining a hypothesis based on previous findings for minimally disturbed sites, we deemed it to be unnecessary to use this approach of subsetting by human influence in any subsequent analyses. The similarity of the results for $\text{NO}_3\text{-N}$ indicated that taking a simpler approach of examining full datasets is adequate for determining whether there are patterns between stream N and alder levels in our collated data in general. Thus, we used all 336 independent sites with summer TN data for examining the relationship between alder and TN,

as well as for comparing our TN data to prospective government-mandated nutrient criteria under the Clean Water Act. To compare TN data to these potential nutrient criteria, we included all proposed N criteria for the OCR on our plot of summer TN data from the following sources: USEPA's draft TN criterion for the Western Forest Mountains nutrient ecoregion from USEPA (2000) and a range of potential TN criteria from Herlihy and Sifneos (2008), a regional survey which calculated potential TN criteria for subregions of the PNW using a variety of data sources and methods. As mentioned above, TN data presented complexities in terms of multiple data sources and high detection limits so we did not do formal hypothesis testing or extensive analyses of TN data. However, the TN data, overall, are sound (see Appendix S1) and important to include in this analysis in comparison to nutrient criteria because management agencies base and evaluate N nutrient criteria using TN data of this quality and complexity (USEPA, 2000; Herlihy and Sifneos, 2008).

Distance to the coast presented a potential confounding factor in the hypothesis test. $\text{NO}_3\text{-N}$ is expected to decrease with distance inland due to lower sea salt inputs (Compton and Church, 2011); however, alder also decreases with distance inland in the OCR (Figure 1) (Ohmann and Gregory, 2002). Thus, we looked at two elements to evaluate whether distance to the coast was confounding a relationship between alder and $\text{NO}_3\text{-N}$. First, we estimated the product-moment correlation (a measure of the correlation or linear dependence between two values) between distance to the coast and basal area of alder (transformed) in our study sites (Burnham and Anderson, 2002). Second, we produced a trellis plot. A trellis plot displays changes in the relationship between two variables as a third covariate varies (Yeager *et al.*, 2007). In this case, we plotted summer $\text{NO}_3\text{-N}$ *vs.* basal area of alder binned by distance to the coast. If distance to the coast is not confounding the hypothesis test, we would expect to see $\text{NO}_3\text{-N}$ showing a consistent change with the GNN alder metric even when binned by distance to the coast; each panel of the trellis plot should show a similar (increasing) relationship between $\text{NO}_3\text{-N}$ and alder. We also examined distance to the coast in our exploratory analyses (see section on Exploratory Analyses of Summer N).

To evaluate whether coarse vegetation data from the NLCD are good surrogates of alder cover across the OCR, we first estimated correlations between the GNN alder metric and the most likely NLCD surrogates of alder cover (percent deciduous forest and percent mixed forest). We also examined relationships between these coarse vegetation data from the NLCD and the full summer $\text{NO}_3\text{-N}$ dataset ($n = 386$

sites), regressing summer $\text{NO}_3\text{-N}$ on percent deciduous forest and percent mixed forest separately as well as on percent deciduous and percent mixed forest together.

Seasonal Analysis of $\text{NO}_3\text{-N}$. To examine seasonal patterns in the relationship between alder and nitrate, we conducted a mixed model analysis on unbalanced calendar season data (R package nlme, maximum likelihood method, Pinheiro *et al.*, 2008). All 482 independent sites with $\text{NO}_3\text{-N}$ data in any of four seasons (defined by calendar-based seasons) were included in mixed models; 321 sites had data in a single season only; 74 had data in two seasons; 19 had data in three seasons; and 68 had data in all four seasons. Seasons were summer (386 sites), fall (135 sites), winter (131 sites), and spring (127 sites). Mixed models included site as a random effect (to model a separate baseline intercept value of $\text{NO}_3\text{-N}$ for every site) and basal area of alder (as a percent of watershed area, arcsine-square root-transformed) as the only continuous predictor and a fixed effect. In our mixed model analysis, we first fit a baseline model of $\text{NO}_3\text{-N}$ vs. the GNN alder metric with no seasonally specific intercepts or slopes. Then, we fit a model that added seasonally specific intercepts, and finally, we fit a full model which included both seasonally specific intercepts and slopes.

Exploratory Analyses of Summer N. We explored relationships between summer $\text{NO}_3\text{-N}$ and additional predictor variables listed in Table 2 in three ways. First, we report simple correlations between all GIS variables and summer $\text{NO}_3\text{-N}$ and summer TN. Second, we conducted an all-subsets regression analysis (a type of general regression model that seeks a ranking of the strength of evidence for all possible models) using GIS variables that were appropriate for such an analysis. In all-subsets regression, models that better fit the data and are more parsimonious have lower values of AIC (Akaike's Information Criterion); thus, we used the leaps package in R and sought the model with the lowest AIC. Several variables could not be included as candidate variables because their lack of variation precluded any chance that they might show an association with summer $\text{NO}_3\text{-N}$ (i.e., these variables do not have a linear relationship with summer $\text{NO}_3\text{-N}$; cf., Burnham and Anderson, 2002). Variables not included as candidates, due to insufficient variation, were watershed-level percent agricultural land cover (only 5% of sites had agricultural land cover greater than 2.5%), number of CAFOs (only 4% of sites had any CAFOs in the watershed), and number of lakes and reservoirs (only 10% of sites had any lakes or reservoirs in the watershed and only 2% had more than one lake or

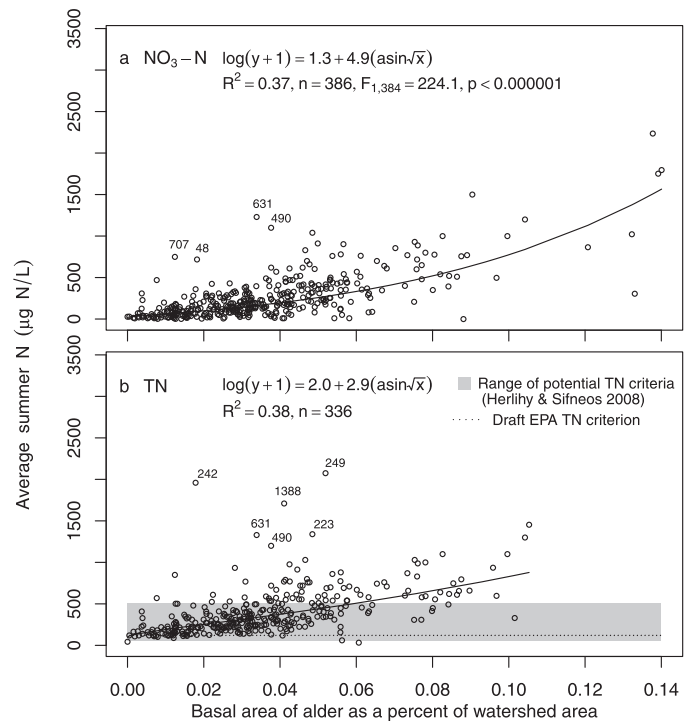


FIGURE 2. Average Summer $\text{NO}_3\text{-N}$, Average Summer TN, and Proposed Nutrient Criteria vs. Alder. The draft U.S. Environmental Protection Agency (USEPA) TN criterion (dashed line) is from USEPA (2000) and represents a criterion calculated for the entire Western Forested Mountains nutrient ecoregion, an area that spans mountainous regions across 13 western U.S. states. Potential TN criteria from Herlihy and Sifneos (2008) range from 55 to 509 $\mu\text{g N/l}$ (gray band). The low end of this range represents the 25th percentile of Environmental Monitoring and Assessment data across the Western Forested Mountains nutrient ecoregion. The high end represents a value for Coast Range, Puget Lowland, and Willamette Valley sites with high cover of hardwoods, determined from a typology of Pacific Northwest streams. Plots include back-transformed regressions of $\log+1$ -transformed $\text{NO}_3\text{-N}$ against arcsine-square root-transformed basal area of alder as a percent of watershed area. Data points identified visually as having low levels of alder but high levels of $\text{NO}_3\text{-N}$ and TN are labeled with an adjacent site ID number.

reservoir). We also dropped site-level land cover type from the candidate variables because it was highly correlated with distance to the coast. Mean distance to the coast at sites with a site-level land cover type of agriculture or urban was 14.7 km, whereas mean distance to the coast at sites with a site-level land cover type of natural was 27.8 km ($t = -4.5$, $df = 47$, $p = 0.00004$). To check whether the correlation between agricultural/urban site-level land cover type and distance to the coast was confounding the effect of distance to the coast indicated in the lowest AIC model (see Results section), we also ran the all-subsets analysis only using sites with a natural site-level land cover type and found that results were the same whether sites with human disturbance at the site level were included or not. We also confirmed that the

model did not include correlated predictor variables using a pairs plot of product-moment correlations among predictor variables (cf., Yeager *et al.*, 2007).

For our third exploration of variables influencing summer NO₃-N, we examined characteristics of sites with low levels of alder but high levels of NO₃-N and TN. When looking at Figures 2 and 5, 10 data points in Figure 2 and three data points in Figure 5 had low levels of alder but high N. These 13 data points represented 11 sites, which are labeled in Figures 2 and 5 with their site numbers.

RESULTS

Alder Influences on Summer N Concentrations

The test of our main hypothesis that alder levels are associated with stream NO₃-N in minimally disturbed sites of the OCR was highly significant ($R^2 = 0.451$, $n = 324$ sites, $F_{1,322} = 264.1$, $p < 0.00001$), indicating that summer nitrate increases with the GNN alder metric as follows in Equation (1):

$$\log\text{NO}_3\text{-N} = 1.28 + 5.09 \cdot \text{asin}(\sqrt{\text{alder}}) \quad (1)$$

When we relaxed our criteria and included summer NO₃-N data from all 386 independent sites, regardless of the amount of natural land cover and CAFOs, the relationship between basal area of alder and NO₃-N was still similar to that in the hypothesis test, with alder accounting for 37% of the variation in summer NO₃-N over all 386 sites (Figure 2a) *vs.* 45% of the variation in summer NO₃-N over the 324 sites we used in the hypothesis test above. Summer TN also showed an increasing relationship with basal area of alder, which accounted for 38% of the variation in summer TN (Figure 2b). Comparison of the TN-alder relationship with proposed nutrient criteria indicate that, with high levels of alder, OCR streams are expected to exceed even the highest proposed criteria (Figure 2b). Distance to the coast did not confound the overall OCR-wide relationship between alder and NO₃-N, as evidenced by patterns in our trellis plot, as well as a low correlation between distance to the coast and basal area of alder. Patterns of increasing NO₃-N with basal area of alder were observed in all classes of distance to the coast (Figure 3), similar to the overall pattern in Figure 2a. The correlation between distance to the coast and basal area of alder (transformed) in our study sites was $r = -0.36$.

Although basal area of alder as a percent of watershed area was highly correlated with both per-

cent deciduous forest ($r = 0.71$) and percent mixed forest ($r = 0.86$), relationships between NO₃-N and coarse vegetation data from the NLCD were not as strong as the relationship between NO₃-N and basal area of alder derived from the GNN map. Separate linear regressions against percent deciduous forest and percent mixed forest showed increases in summer NO₃-N with both of these variables (Figure 4), but the relationships between NO₃-N and these NLCD data had lower R^2 values than did the relationship between NO₃-N and the GNN alder metric (compare R^2 values of Figures 2a and 4). Regression against both percent deciduous and percent mixed forest likewise showed an increase in summer NO₃-N with these NLCD data, as follows in Equation (2):

$$\begin{aligned} \log\text{NO}_3\text{-N} = & 1.64 + 1.12 \cdot \text{asin}(\sqrt{\% \text{deciduous}}) \\ & + 0.93 \cdot \text{asin}(\sqrt{\% \text{mixed}}) \end{aligned} \quad (2)$$

With an R^2 of 0.28 ($n = 386$), the relationship in Equation (2) had a higher R^2 than did the separate regressions, but it was still low compared to that against the GNN alder metric.

Seasonal Analysis of NO₃-N

In our mixed model analysis of NO₃-N *vs.* alder in all four calendar seasons, the model adding seasonally specific intercepts showed an effect of season on log-transformed NO₃-N (likelihood ratio = 150.3, $df = 3$, p -value < 0.0001), showing that inclusion of seasonally specific intercepts better fit the data than did the baseline model of log-transformed NO₃-N *vs.* the GNN alder metric without any seasonally specific intercepts or slopes. Further adding seasonally specific slopes to the model with seasonally specific intercepts indicated no interaction between alder and season (likelihood ratio = 1.4, $df = 3$, p -value = 0.7). Since this analysis indicated that the baseline model and the model adding seasonally specific slopes were not the best fit for the seasonal data, we do not present these models in detail. The best model fit seasonally specific intercepts and showed that NO₃-N increases with the GNN alder metric in all four seasons (Figure 5). This model also indicated that NO₃-N was highest in winter, comparable in fall and spring and lowest in summer (Figure 5), at any fixed level of alder. In addition to showing that high-flow seasons (winter, fall, and spring) have higher NO₃-N than does the low-flow season (summer), the back-transformed model predictions show that the difference between NO₃-N in high-flow seasons *vs.* in the summer low-flow season increases with increasing levels of alder (Figure 5).

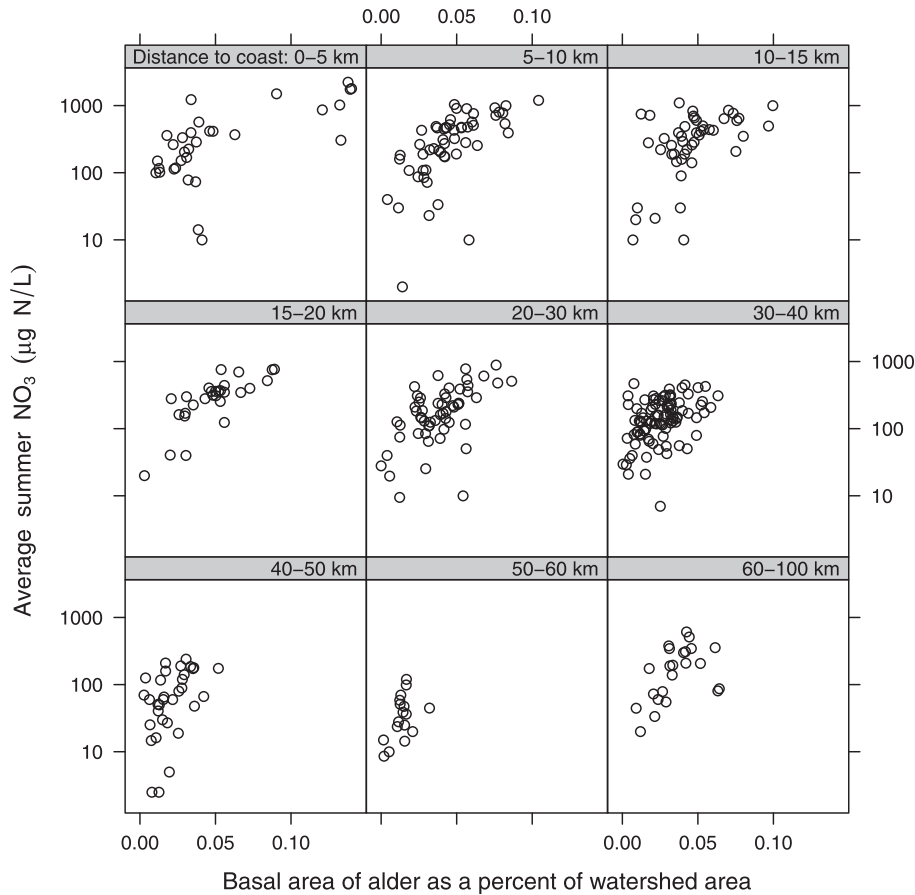


FIGURE 3. Average Summer $\text{NO}_3\text{-N}$ vs. Alder Binned by Distance to the Coast. Trellis plot (cf., Yeager *et al.*, 2007) of average summer $\text{NO}_3\text{-N}$ plotted against basal area of alder binned by distance to the coast.

Exploratory Analyses of Summer N

Simple correlations between all GIS variables and summer $\text{NO}_3\text{-N}$ and TN are listed in Table 2; only the correlations between basal area of alder as a percent of watershed area and N data are higher than 0.5. Of several nearly equal models in the all-subsets regression utilizing the subset of these GIS variables that were appropriate to include, the model with the lowest AIC had an R^2 of 0.46:

$$\begin{aligned} \log\text{NO}_3\text{N} = & -3.884 + 3.869 \cdot \text{asin}(\sqrt{\text{alder}}) \\ & + 0.219 \cdot \text{latitude} - 0.005 \\ & \cdot \text{distance to coast} \\ & - 0.087 \cdot \log(\text{watershed area}) \end{aligned} \quad (3)$$

Note that units for variables in Equation (3) are listed in Table 2. The four predictor variables (basal area of alder as a percent of watershed area, latitude, distance to coast, and watershed area) in this lowest AIC model have approximately equal importance because all were included in all 15 models having

AIC within 10 units of the minimum (Burnham and Anderson, 2002). In the pairs plot (data not shown), the highest absolute value for a correlation among the four predictor variables was between distance to the coast and the GNN alder metric, with an $r = -0.36$, indicating that multicollinearity is not present in this model. Four models had an AIC within 2 units of the lowest AIC model, each of which added one of the other four variables (basal area of other hardwoods as a percent of watershed area, watershed-level percent natural land cover, elevation, and watershed-level percent developed land cover) included in the set of candidate variables entered into the all-subsets analysis. These four models had R^2 values, and regression coefficients for the variables in the lowest AIC model, that were very similar to those of the lowest AIC model. All five of the models had the same R^2 value; the R^2 for each of the five models was 0.46. Ranges in regression coefficients for basal area of alder as a percent of watershed area, latitude, distance to the coast, and watershed area were 3.83 to 3.88, 0.129 to 0.142, -0.006 to -0.005 , and -0.091 to -0.086 , respectively. The regression coefficient for

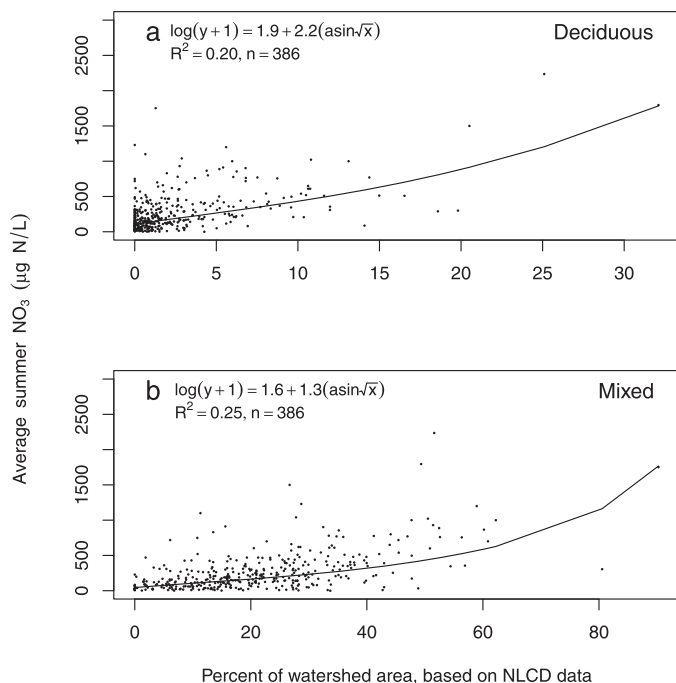


FIGURE 4. Average Summer $\text{NO}_3\text{-N}$ Plotted Against 2001 National Land Cover Dataset (NLCD) Data. NLCD (Homer *et al.*, 2004) data are percent deciduous forest (a) and percent mixed forest (b). Plots include back-transformed regressions of $\log+1$ -transformed $\text{NO}_3\text{-N}$ against arcsine-square root-transformed NLCD data.

basal area of other hardwoods (transformed) in the second-lowest AIC model was +0.292; basal area of other hardwoods was moderately correlated with latitude ($r = 0.6$). The regression coefficient for watershed-level percent natural land cover (transformed) in the third-lowest AIC model was +0.08. The regression coefficient for elevation (transformed) in the fourth-lowest AIC model was very low at +0.0003; elevation was moderately correlated with basal area of alder ($r = 0.59$). Finally, the regression coefficient for watershed-level percent developed land cover (transformed) in the fifth-lowest AIC model was low at +0.001.

Ten of 11 sites with low levels of alder but high levels of $\text{NO}_3\text{-N}$ and TN (labeled in Figures 2 and 5) had watershed-level percent developed land cover and/or percent agricultural land cover which were higher than the medians in Table 2, and the one that did not fit this characteristic had a site-level land cover type of agriculture (also see Table 4 in Appendix S1). In addition, 10 of 11 low-alder/high-N sites had a distance to the coast that was below the median distance to coast in Table 2 (also see Table 4 in Appendix S1). None of these points were influential, as indicated by Cook's distance (an estimate of the influence of a data point) being less than 0.5 for all 11 points (cf., Herlihy and Sifneos, 2008).

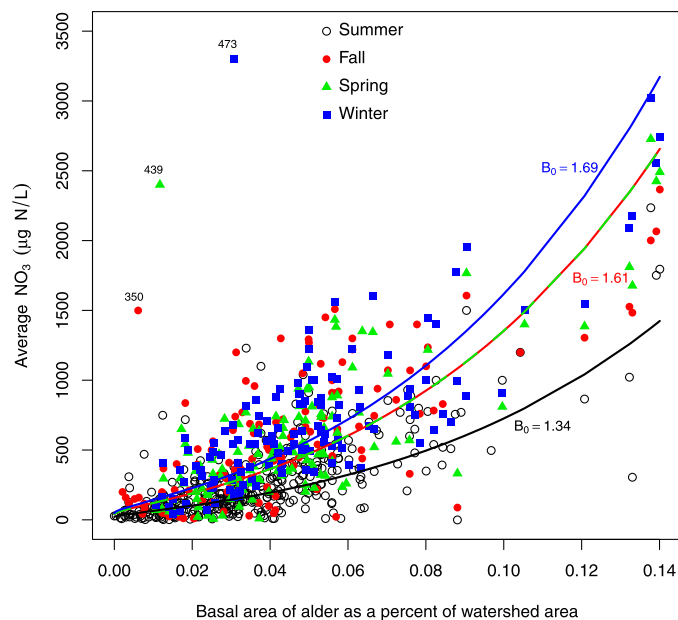


FIGURE 5. Average $\text{NO}_3\text{-N}$ in Calendar-Based Seasons *vs.* Alder. Plot is of all 482 independent sites with $\text{NO}_3\text{-N}$ data in any of four seasons: summer ($n = 386$), fall ($n = 135$), winter ($n = 131$), and spring ($n = 127$). Curves show predictions from the best mixed model: $\text{NO}_3 - \text{N} = 10^{(C_i + B_0 + 4.72 \cdot \text{asin}(\sqrt{\text{alder}}))}$, for seasonally different values of B_0 ; the top curve plots winter, the middle dashed curve plots fall and spring, and the bottom curve plots summer. The random effect of site i (C_i , standard deviation = 0.267) is set to its mean value (mean = 0), so curves express model predictions for an “average” site. Data points identified visually as having low levels of alder but high levels of $\text{NO}_3\text{-N}$ are labeled with an adjacent site ID number.

DISCUSSION

High Stream Nitrogen in the Oregon Coast Range

OCR streams have remarkably high concentrations of N for forested systems (Wigington *et al.*, 1998), and these appear to be due to red alder and possibly to sea salt inputs. The LEMMA group's detailed vegetation mapping using the GNN method is a powerful tool, which allowed us to determine an alder-specific metric for watersheds across the OCR and conduct a strong test of the hypothesis of Wigington *et al.* (1998): spatial variation in stream $\text{NO}_3\text{-N}$ across the OCR is influenced by watershed alder cover. The criteria for including sites in the main hypothesis test were based on characteristics of sites in the Salmon River basin, the 200-km² OCR drainage in which a surrogate of alder cover (hardwood cover from an earlier version of an OCR GNN map) was previously shown to correlate with stream $\text{NO}_3\text{-N}$ across 18 sites (Compton *et al.*, 2003). Our test of alder influences on stream N regressed stream $\text{NO}_3\text{-N}$ against a direct measure of alder using 324 sites over the entire

52,000-km² OCR region. Furthermore, the use of hardwood cover as a surrogate for alder by Compton *et al.* (2003) worked well because 90% of hardwood cover in the small Salmon River basin is alder, but across the entire region, hardwood cover is not universally dominated by alder; other hardwoods predominate in southern and western parts of the OCR. Thus, the mapping of the basal area of single tree species is a key innovation of the GNN mapping method; this allows us to scale up the alder-N relationship to the entire region by looking only at alder.

Our findings indicating red alder's role in producing high N in OCR streams are supported by recent research across the PNW. Herlihy and Sifneos (2008) found a significant relationship across Oregon, Washington, and Idaho between N and hardwood forest cover, a relationship that is thought to be attributable in part to red alder in the PNW's hardwood forests. Updates to USGS SPARROW (Spatially Referenced Regressions on Watershed Attributes) models for the PNW also indicate the importance of red alder as a stream N source in this region. Previous SPARROW models, which lacked specific forest vegetation metrics such as red alder cover, produced a cluster of underpredicted N (see figure 4 in Smith *et al.*, 1997) in the Oregon and Washington Coast Ranges, both areas where red alder is abundant. Based on previous work (Compton *et al.*, 2003), Wise and Johnson (2011) included a red alder metric in addition to forest land cover, and were able to produce more realistic SPARROW models for the PNW (see figure 2 in Wise and Johnson, 2011). Of the 178 sites used to produce their PNW-wide models, only five were Coast Range sites in Oregon; however, for streams of the Oregon and Washington Coast Ranges, estimates by Wise and Johnson (2011) indicate that red alder contributes one-quarter to one-half of the TN load and is the dominant source of TN.

Distance to the coast and latitude, which are likely and possible surrogates, respectively, for sea salt inputs to watersheds, were also related to summer nitrate values in our exploratory analysis. Sea salt (Wigington *et al.*, 1998) and nitrate (our analysis) concentrations both decrease with increasing distance from the coast. Likewise, we found a definite correlation between summer nitrate and latitude, and sea salt inputs may increase with latitude. Precipitation increases with latitude along the Oregon coast (PRISM Climate Group, 2006), and storms originating from the Pacific Ocean may be more likely to make landfall at higher latitudes within the OCR; thus, atmospheric and flood-based inputs of sea salts to land may also be greater at these higher latitudes (Robbins Church, USEPA Western Ecology Division, April 15, 2008, personal communication). Experiments on OCR

soils indicate that additions of sodium chloride in sea salts at concentrations similar to those found in salt deposition stimulate nitrate leaching from soils (Compton and Church, 2011). Future research should further examine these sea salt hypotheses, for both distance inland and latitude, with data on streamwater chloride and/or chloride deposition data. Examining whether chloride deposition does in fact vary with latitude would be ideal in evaluating the hypothesis that NO₃-N increases with latitude due to sea salt inputs, but the only deposition monitoring site in the OCR is currently inactive with a period of record from 1979 to 2007 (NADP, 2000).

In the lowest AIC model from the exploratory analysis, smaller watershed areas were also associated with higher NO₃-N concentrations, a finding previously reported for streams in the OCR (Wigington *et al.*, 1998) and possibly explained by greater algal uptake of NO₃-N in larger streams (Wall *et al.*, 1998; Binkley *et al.*, 2004). In contrast, we were not able to fully examine possible relationships between land uses such as agriculture, CAFOs, and urban development and high N in OCR streams. This is likely because lowland tidal streams and estuaries where these human influences are most commonly located in the OCR were not included in our analyses. Watershed-level percent agricultural land cover and number of CAFOs were not variable enough in our sites to show an effect on summer NO₃-N in the all-subsets analyses. Likewise, there is little developed land cover in non-tidal streams of the OCR (90% of our summer NO₃-N sites had watershed-level percent natural land cover greater than 90%), and this dominance of forested and natural land cover may explain why watershed-level percent developed land cover was not in the lowest AIC model. However, characteristics of low-alder/high-N sites (i.e., labeled sites in Figures 2 and 5) indicate that OCR watersheds with high agricultural and developed land cover can have elevated N levels. Proximity to the coast may be an alternate or additional explanation of the high N values at these low-alder/high-N sites. These findings are similar to the findings of Herlihy *et al.* (2005). Their examination of human activity and a macroinvertebrate-based index of biotic integrity (IBI) in OCR headwater sites also indicated that the few moderately and severely impaired sites were influenced by agricultural land cover.

There are a variety of anthropogenic activities in OCR streams/watersheds that we did not examine for relationships with stream N. These included salmon carcass additions (Compton *et al.*, 2006; Wise and Johnson, 2011), roads (Spies *et al.*, 2002), forest fertilization (Sullivan *et al.*, 2005), as well as the most widespread human activity affecting the OCR landscape: current and past logging (Naymik *et al.*, 2005).

For salmon carcass additions, roads, and forest fertilization, appropriate GIS layers were not available. The complexity in the OCR of both the available GIS forest harvest and disturbance history data (Lennartz, 2005) and reported relationships between N and logging (Wigington *et al.*, 1998) put forest harvest and successional status outside the scope of our analysis but should be considered in future regional analyses that build on this one. Although GIS data limitations precluded a thorough examination of all anthropogenic activities, we were able to examine NLCD vegetation data that researchers may try to use as surrogates for alder cover in comparison to the GNN alder metric. Deciduous and mixed forest covers from the NLCD map were less predictive of stream nitrate than the GNN alder metric was. This may be due to NLCD deciduous and mixed forest covers including hardwoods other than alder. NLCD deciduous and mixed forest also does not account for any sparse alder occurring in parts of the watershed that the NLCD map characterizes as evergreen, whereas the GNN alder metric provides the basal area of alder that occurs in watersheds and plots dominated by conifers.

Seasonal patterns in $\text{NO}_3\text{-N}$ concentrations across the OCR (Figure 5) are similar to others' findings for OCR streams if we apply calendar seasons to their data (e.g., highest in winter, lowest in summer, and ranging from low to high in fall and spring, Colbert and McManus, 2003; Sullivan *et al.*, 2005; Poor and McDonnell, 2007; Sigleo and Frick, 2007). A start date for the fall season coinciding with the first fall flush in October or November (when the first storm following the low flows of August and September typically occurs in OCR streams, e.g. Sigleo and Frick, 2007) rather than the calendar start date for fall (~September 23) would have been more likely to yield fall as the season with the highest $\text{NO}_3\text{-N}$ concentrations, a result we expected based on above citations, which had sufficient hydrology data to define a hydrology-based definition of fall. However, the appropriate start date for a hydrology-based definition of the fall season probably varies across years and across the OCR with latitude, necessitating a dataset and analysis of hydrology which we did not have. Our results showed not only that high-flow seasons (winter, fall, and spring) had higher $\text{NO}_3\text{-N}$ than did the low-flow season (summer), but that the difference between $\text{NO}_3\text{-N}$ in high-flow seasons *vs.* in the summer low-flow season increases with increasing levels of alder. This pattern is consistent with findings for other Oregon streams (agricultural streams in the Willamette Valley, Pan *et al.*, 2004) that relationships between landscape properties and stream condition are stronger during periods of higher flow. In addition, biological uptake of N can also be

influenced by temperature and season (Mulholland, 2004), possibly further contributing to low stream N concentrations in the summer. Our seasonal results further illustrate findings from Pan *et al.* (1996), another analysis based on EMAP data. Monitoring programs, such as EMAP, which only collect data once per site during summer low-flow conditions provide only a snapshot of stream nutrient conditions and their relationships with landscape properties. Our inclusion of data from high-flow seasons in the seasonal analysis confirmed that $\text{NO}_3\text{-N}$ increases with alder in other seasons, as it does in summer. Data from high-flow seasons also added new information by showing that relationships between alder and $\text{NO}_3\text{-N}$ were stronger in winter, fall, and spring. By only monitoring the lowest $\text{NO}_3\text{-N}$ concentrations of the year, summer-based monitoring programs may understate $\text{NO}_3\text{-N}$ associations with landscape properties such as alder, especially in streams with strong seasonal patterns in flow.

Implications for Nutrient Management

Alder and sea salt sources of high N in OCR streams present a challenge to developing and applying nutrient criteria. Our evidence for the challenge that alder presents is fairly strong, whereas the role of sea salt in causing high N levels remains speculative. USEPA's draft criterion (120 $\mu\text{g N/l}$, USEPA, 2000) for Western Forested Mountains is particularly low compared to TN data in our analysis, with TN exceeding this level across a wide range of alder cover (Figure 2). This draft criterion is widely regarded, including in its original publication, as unlikely to apply universally across such a large area (Ice and Binkley, 2003). However, even in a regional survey, by Herlihy and Sifneos (2008), focused specifically on parsing out regions within the PNW, the highest proposed criterion (509 $\mu\text{g N/l}$) calculated for high hardwood-cover reference watersheds in the Coast Range was lower than many high alder-cover streams in our analysis (72% of watersheds in Figure 2b with percent basal area of alder above 0.06 had TN greater than 509 $\mu\text{g N/l}$). Compared to our analysis of 336 sites, Herlihy and Sifneos (2008) used 36 OCR streams in their data-based typology of PNW reference streams. If this data-based typology of PNW reference streams were run with a larger number of OCR sites and/or sites with the highest levels of alder cover, the inclusion of a wider range of alder cover might lead to a higher proposed criterion for hardwood-dominated Coast Range watersheds.

Because criteria are often based on reference conditions, determining criteria can be difficult when regions include areas with natural sources of

high-nutrient loads. Naturally high-nutrient streams might be inappropriately listed as impaired under the Clean Water Act if nationally and regionally developed criteria are applied to them (Ice and Binkley, 2003; Compton *et al.*, 2006). On the other hand, areas with naturally high nutrients may require stricter nutrient controls because ecosystems may be on the brink of nutrient-related problems with even small additions of nutrients from human activities (cf., Todd *et al.*, 2009). Human activities in the OCR which either do or have potential to add various levels of N to streams include forest harvest, forest fertilizations, CAFOs, and other agriculture, and current as well as future expected increases in residential land development (Johnson *et al.*, 2007).

Criteria development is further complicated when there is debate over what constitutes appropriate reference conditions. Reference sites for determining natural background conditions are often nonexistent, leading to questions about whether to estimate natural conditions by correcting for anthropogenic influences or to use least-impacted conditions as (“unnatural”) background conditions (Smith *et al.*, 2003). Alder in the OCR provides a potential example. Alder is a native pioneer species, which occurred across the landscape historically, colonizing areas impacted by natural disturbances such as fires, floods, avalanches, and landslides (Long *et al.*, 1998; Long and Whitlock, 2002). However, currently widespread alder distributions are legacies of human disturbances including harvest, burning, land clearance, and alterations in the spatial and temporal scales of natural disturbance regimes (Hibbs *et al.*, 1994). How current distributions compare with historical distributions prior to intensive landscape alterations associated with European settlement remains unclear (Ohmann *et al.*, 2007). OCR alder cover is thought to have increased during 19th and early 20th Century land clearance and forest harvest practices, whereas more recent forest management practices have caused declines in hardwood cover compared to high levels in the 1930s (Kennedy and Spies, 2005). Thus, high stream N concentrations in alder-dominated basins may be equivalent to natural/undisturbed conditions (Ice and Binkley, 2003), yet current levels of alder-derived N may often be attributable as well to anthropogenic disturbance.

Evidence that alder-derived N can contribute to impaired stream conditions comes from other studies. First, IBI scores based on diversity and abundance of aquatic vertebrates are negatively correlated with N and P in the Oregon and Washington Coast Ranges (Kaufmann and Hughes, 2006). Second, OR DEQ has suggested that nitrate is the main factor limiting water quality in the Salmon and Yaquina River basins (Cude, 2006), two OCR watersheds where

USEPA researchers have identified alder as the primary source of stream N (Compton *et al.*, 2003; Sigleo and Frick, 2007; Brown and Ozretich, 2009). However, high nitrate in these systems also occur during heavy precipitation and are accompanied by increased phosphorus, total solids, and biochemical oxygen demand, and it remains unclear whether these water quality characteristics are attributable to natural alder sources *vs.* other sources of nonpoint source pollution (Cude, 2006). Our work suggests that the widespread presence of alder could play a role in the potential impairment of these waters.

Examination of existing data and the establishment and evaluation of proposed nutrient criteria in Oregon generally may also be limited by methodological issues. The detection limit for OR DEQ’s TKN method is 200 $\mu\text{g N/l}$ and is above many of the proposed TN criteria for the region (cf., prospective criteria in Herlihy and Sifneos, 2008 and 120 $\mu\text{g N/l}$ criterion in USEPA, 2000). This detection limit is also above the TKN levels in many OCR streams. For example, almost two-thirds of the TKN values from OR DEQ in our TN analysis were below this detection limit, and estimates of TN based on this below detection TKN data represented roughly half of the summer TN data we found for the OCR. This high detection level limited the types of analyses we were able to do for TN. For example, we could not examine the proportion of TN which was $\text{NO}_3\text{-N}$, as has been done by other researchers conducting regional analyses (e.g., Stanley and Maxted, 2008). Moreover, either TN or estimates of TN based on TKN plus nitrate are the recommended analytes to use for numeric N criteria (USEPA, 2000), yet OR DEQ’s only available TN/TKN plus nitrate data are based on a TKN analysis that has a detection limit higher than the majority of both proposed TN criteria and typical mountain stream TN values. Thus, these statistics indicate that the majority of data currently collected by the state agency that is responsible for establishing water quality criteria in Oregon are inadequate for both setting numeric N criteria in Oregon’s mountain regions and determining whether streams meet those criteria.

There are also challenges to setting nutrient criteria in the PNW because of the predominance of forested stream systems, in which relationships between salmon and nutrients are a major focus of management but not research (Compton *et al.*, 2006). Research on nonpoint sources of nutrients, and efforts to protect designated uses from nutrient pollution, have focused on autotrophic systems where nutrients stimulate harmful algal blooms and hypoxia. Such effects are unlikely when algal growth is limited by the low light levels and hydrologic scouring typical of many forested systems (Stockner and

Shortreed, 1978; Biggs, 2000). However, altered nutrient loads to forested streams can interact with changes in light levels and hydrology due to forestry practices, fire, and other natural and anthropogenic disturbance regimes, and these nutrients may be transported downstream to larger streams and estuaries where algal blooms are not light limited (Kentula and DeWitt, 2003). Altered nutrient dynamics in forest streams can also affect detrital food webs (Cross *et al.*, 2006). However, comparatively little research has been done on consequences of increased stream nutrients in heterotrophic streams, and efforts to develop nutrient criteria do not seem to consider the potential for eutrophication of nonautotrophic systems (Dodds, 2007). Abundance of Coho salmon in the OCR's West Fork Smith River basin show correlations with TN (Ebersole *et al.*, 2009), potentially reflecting salmonid dependence on nutrient status of detritus-based food webs. Thus, the potential effects of nutrients on designated uses in forested regions remains unclear, and this is particularly true for the OCR (Binkley *et al.*, 2004).

Depending on how management agencies view nutrients that leach from alder stands into streams (natural background nutrients *vs.* due to human disturbance), our models, or similar ones, could be useful to setting nutrient criteria in nontidal streams of the OCR. If agencies view alder as a natural background source of nutrients, a multiple regression model, like our exploratory model for summer nitrate, might be used to set site-specific criteria (i.e., by setting the model to the site's specific level of alder, distance to the coast, latitude, and watershed area). Or a multiple regression model might be used to moderate a low OCR-wide N criterion. Sites which are listed due to this OCR-wide criterion might be delisted if a multiple regression model indicates that the high N levels are due to the site's watershed alder, distance to the coast, latitude, and watershed area. Alternatively, if agencies view alder as natural in some stands and unnatural, due to human disturbance, in other stands, forest vegetation models or information on human disturbance history at the site could be combined with a multiple regression model of N. The multiple regression model could be used to set a criterion by setting the equation to the natural level of alder expected at the site, the distance to coast, latitude, watershed area, and natural levels of other variables included in future models. More research on what constitutes natural and unnatural distributions of alder in the OCR would be necessary for such an approach.

Nutrient criteria development requires an understanding of natural and anthropogenic-related variation in nutrients as well as their variable relationships with landscape characteristics across

different regional spatial scales (Rohm *et al.*, 2002). Our results indicate that at the scale of the OCR, streams with higher levels of alder in the watershed tend to have higher N, often above numeric N criteria proposed from analyses at much larger national scales (USEPA, 2000) and at comparable regional scales (Herlihy and Sifneos, 2008). The important role that red alder plays in the biogeochemistry of OCR streams must be considered in aquatic ecosystem management of the region.

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Appendix for "Linking Landscape Characteristics and Stream Nitrogen in the Oregon Coast Range: Red Alder Complicates Use of Nutrient Criteria"—E.A. Greathouse, J.E. Compton, and J. Van Sickle.

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