City of Bellevue

DATE: October 28, 2009

TO: Chair Sheffels, Members of the Planning Commission

FROM: Michael Paine, Environmental Planning Manager, DSD, 452-2739
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SUBJECT: Shoreline Master Program (SMP) Update—Bellevue File # 07-122342 AC

INTRODUCTION

At the October 28th study session, Tessa Francis will present a brief overview of her research on the effects of shoreline urbanization and aquatic ecosystems. She will also describe results from a study into the collection and incorporation of scientific information by local jurisdictions in Western Washington as part of their GMA-mandated update of critical areas regulations. Ms. Francis graduated from the University of California at Berkeley and received her PhD in Zoology and Urban Ecology from the University of Washington. She is currently a National Research Council Postdoctoral Fellow at NOAA's Northwest Fisheries Science Center. This study session is the third in a series of science and technical presentations aimed at providing the Commission and the public with background information on key issues relevant to the Update process.

EFFECTS OF SHORELINE URBANIZATION ON NORTHWEST LAKES

Ms. Francis will present data from a study of 25 lowland Pacific Northwest lakes describing how shoreline development interrupts variable inputs of terrestrial material into lakes, and the associated consequences for aquatic habitats and food webs. The details of her presentation are outlined below.

"Humans preferentially settle near freshwaters, including along the shorelines of lakes. Because of the tight coupling between riparian and aquatic habitats, it is likely that this human disturbance alters terrestrial-aquatic linkages on lakes, impacting species interactions and ecosystem function. Here I will present data collected from 25 lowland Pacific Northwest lakes, along a gradient of shoreline development, describing how shoreline development interrupts fluxes of terrestrial material to lakes, and the associated consequences for aquatic habitats and food webs. The density of riparian trees and woody debris, and the organic content of littoral sediments, decline significantly with shoreline development. I show that coarse wood is a physical structure that retains organic matter in shallow waters where it is available for biotic processing, and that the loss of organic matter in urban lake sediments is associated with reduced benthic invertebrate diversity. I also show that riparian deforestation on urban lakes is associated with a loss of terrestrial insects from fish diets, a result confirmed at multiple sampling scales: a focused

comparison of two urban lakes and two lakes with intact riparian forests, a one-time sampling survey of 28 Pacific Northwest lakes, and a literature review of 25 North American lakes. In addition, fish in undeveloped lakes daily consume up to 3x the amount of energy consumed by fish in urban lakes. These results highlight the importance of aquatic-terrestrial linkages for lake habitat structure and food web interactions, and suggest that, as shorelines are developed, retention of riparian vegetation and littoral coarse wood is critical for maintaining food web connections.

As requested by the Commission, copies of three relevant peer reviewed papers by Ms Francis and her contributory authors are included as Attachments 1, 2 and 3.

MEETING QUESTION FORMAT

As previously introduced at the last study session, staff will again be distributing note cards to the audience at the beginning of the presentation. If a member of the audience has a question specific to the presentation, they should write the question on the note card provided. At the end of the presentation, the note cards will be collected and delivered to the Commission Chair. After the Commissioners have had a chance to ask questions of the presenter, the Chair will select a few questions from the audience that are relevant to the presentation by the guest speaker. Questions on alternate topics outside the scope of the presentation, including questions from the Commission on detailed material previously presented to the Commission, should generally be reserved to a later date so as to ensure appropriate use of the speaker's time.

We anticipate this presentation to last approximately 45 minutes, and have allowed for approximately 15-20 minutes for questions from the Commission and the public following the presentation. The Commission will be using this process to manage questions from the public directed to the speaker throughout the speaker series.

ISSUES LIST

As identified in the September 23, 2009 agenda memo, staff has compiled a list of questions previously raised by the Commission in past meetings. For the Commission's convenience, staff is tracking each item and, on this most recent version (see desk packet), has identified graphically whether an item has been addressed or, if the item is still pending, when the anticipated resolution will occur. This improved version has been published to the SMP Update web site. No additional questions were identified at the last meeting.

SCIENCE DOCUMENTS

The Commission has requested a number of documents used during Planning Commission's past review of proposed changes to the critical areas ordinance (CAO) which culminated in a Council adopted ordinance effective August, 2006. These include the GMA Best Available Science document for critical areas (including the Shoreline Area) and the Critical Areas Risk Analysis

(which assessed overall risk to ecological functions from two alternatives—and no action—under consideration at the time.) There are also two studies produced by the City of Bellevue and titled respectively *A Summary of the Effects of Bulkheads, Piers, and Other Artificial Structures and Shoreline Development on ESA-listed Salmonids in Lakes* and *Lake Sammamish Ordinary High Water Mark Study—Final Report.* The bulkhead study was commissioned by the city following the 1999 ESA listing of Puget Sound Chinook. The high water mark study was designed by the City to provide residents and staff consistent average elevation for establishing setbacks for residential development from OHWM. We have also incorporated correspondence between the City and the Department of Ecology regarding Ecology's definition of the ordinary high water. And finally, there is a recent Department of Fish and Wildlife publication titled *Land Use Planning for Salmon, Steelhead and Trout: A Land Use Planner's Guide to Salmonid Habitat Protection and Recovery.* All of these documents will be provided to the Commission via mail, at the meeting, or electronically, per each member's preferences. Links to all of these documents are available on the Shoreline Update web page.

NEXT STEPS

Staff has included a revised schedule with the list of topics and presenters anticipated in the near future as Attachment 4 to this memo. As staff is able to secure additional speakers for scheduled meetings, we will revise the proposed schedule. Where possible, we will try to provide a summary of the speakers proposed presentation in advance of the meeting as part of the published agenda packet.

Research

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Shoreline urbanization reduces terrestrial insect subsidies to fishes in North American lakes

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Despite growing recognition of the energetic connections between aquatic and riparian habitats of streams and lakes, there have been few efforts to quantify the importance of terrestrial insect subsidies to fish in lakes. Further, it is unclear whether lakeshore urbanization alters the magnitude of these fluxes. Because lakeshore development has been found to be negatively correlated with riparian vegetation that serves as habitat for terrestrial invertebrates, we expected that shoreline urbanization would reduce the prevalence of terrestrial invertebrates in fish diets. We quantified the effects of lakeshore urbanization on terrestrial insect subsidies to fish at three scales: a focused comparison of annual patterns in four lakes in the Pacific Northwest, a one-time field survey of 28 Pacific Northwest lakes, and a literature survey of 24 North American lakes. At all geographical scales, terrestrial invertebrate subsidies to fish were negatively correlated with shoreline development. Terrestrial insects comprised up to 100% of fish diet mass in undeveloped lakes, versus an average of 2% of fish diet mass in developed lakes. Trout, *Oncorhynchus* spp., in undeveloped lakes had an average of 50% greater daily energy intake, up to 50% of which was represented by terrestrial prey. Temporal variability of the terrestrial subsidy suggests that these inputs are distinctly pulsed, and this subsidy is absent or temporally rare in undeveloped lakes.

Improving our understanding of the dominant energy sources and pathways supporting upper trophic levels in aquatic ecosystems has been a focus of ecology for decades (Lindeman 1942, Odum 1968). Traditional models of lake food webs focus on energy flow through pelagic pathways (Carpenter et al. 1987, Brett and Goldman 1997), describing carbon fixed by phytoplankton, and transferred to zooplankton and eventually top consumers such as fish. Observations of omnivory by fishes, including opportunistic and ontogenetic prey switching (Werner and Gilliam 1984, Hodgson and Kitchell 1987), expanded traditional views of strictly pelagic lake food webs to include benthic organisms (Diehl 1992). Lake food web studies are now more commonly informed by conceptual models of energy transfers linking pelagic consumers, benthic prey, benthic primary producers and detrital energy pathways (Schindler and Scheuerell 2002, Vadeboncoeur et al. 2002). Substantial reliance on zoobenthos, both directly and indirectly, is currently recognized as widespread and common to most fish in temperate lakes (Schindler and Scheuerell 2002, Vander Zanden and Vadeboncoeur 2002).

Energy flux between terrestrial and aquatic ecosystems is increasingly recognized as an important component of food webs across a diverse range of biomes (Polis et al. 2004). In lotic ecosystems, tight associations between aquatic and riparian habitats result in prey exchanges across stream-riparian boundaries that subsidize both aquatic and riparian food webs (Power 2001, Sabo and Power 2002), In particular, terrestrial insects using riparian vegetation as habitat can be an important food source for fish in streams (Nakano and Murakami 2001), representing up to half the total energy budget for some fish (Baxter et al. 2005).

Lakes also receive substantial inputs of terrestrial organic matter from their catchments and this carbon pool is fundamental to lake ecosystem function (Jansson et al. 2007). Most evidence of the transfer of terrestrial carbon through limnetic food webs is via the microbial loop (Jansson et al. 2007); examples of the importance of particulate organic matter (POM) from terrestrial habitats are rarer (but see Richey and Wissmar 1979, France and Peters 1995). Though terrestrial organisms are commonly found in fish diets (Hodgson and Hansen 2005), there has been limited systematic quantification of the degree to which fishes in north temperate lakes rely on terrestrial prey sources (but see Mehner et al. 2005).

The human urban population is increasing across North America and the earth in general (United Nations 2007), and human development is concentrated around freshwaters (Walsh et al. 2003). Development of shorelines is associated with a suite of changes to lake habitat structure and ecosystem function, including riparian deforestation (Christensen et al. 1996, Francis and Schindler 2006), loss of coarse wood (Christensen et al. 1996, Francis and Schindler 2006, Marburg et al. 2006) and emergent

vegetation (Jennings et al. 2003) from littoral zones, and eutrophication (Schindler 2006). These human perturbations are associated with various changes in fish ecology, including reduced growth and abundance (Schindler et al. 2000, Sass et al. 2006), and modified foraging behavior (Sass et al. 2006) and spatial distributions (Scheuerell and Schindler 2004). Despite the widespread loss of riparian vegetation associated with shoreline urbanization, there have been no comprehensive studies of how these changes to lake riparian habitats affect the magnitude of terrestrial insect subsidies to aquatic food webs.

We quantified the effects of lakeshore development on terrestrial insect subsidies to fish in north temperate lakes by comparing fish diets in lakes along a gradient of shoreline development. We expected that terrestrial prey would represent a substantial portion of fish diets on lakes with intact shorelines, owing to riparian vegetation fringing lakeshores that serves as habitat for terrestrial organisms. Because human development of shorelines is associated with reduction in the density of riparian vegetation (Francis and Schindler 2006), we further predicted that inputs of terrestrial insects would be reduced on urban lakes as reflected in fish diets. We consider the relative importance of terrestrial insects through a focused sampling of four lakes, then broaden the scale of our investigation with a survey of 28 Pacific Northwest lakes, and a literature survey of an additional 24 North American lakes.

Methods

Focused diet comparison

To assess the effects of shoreline development on terrestrial subsidies to fish, we compared fish diets in two undeveloped lakes (Eunice, Gwendoline) to two developed lakes (Shady, Star) in the Pacific Northwest region of the United States. Eunice and Gwendoline lakes are located in the Univ. of British Columbia's Malcolm Knapp Research Forest near Maple Ridge, British Columbia (19°49′N, 34°122′W) and have fully intact riparian forests. Shady and Star lakes are located in suburban areas of King County, Washington (Shady: 25°47′N, 6°122′W; Star: 21°47′N, 17°122′W), and each has 95% of its shoreline developed by residential dwellings. The four lakes have an average surface area of 13.5 ha (±2.0 SE) and an average depth of 10.8 m (±2.3 SE). The lakes are all located below 500 m elevation in the western hemlock *Tsuga heterophylla* zone of the Cascade Range and Puget Trough regions. Cutthroat trout *Oncorhynchus clarkia* were introduced to Eunice Lake in a transplant experiment in 1974 (Hume and Northcote 1985), and subsequently colonized Gwendoline Lake through tributaries. Both Shady and Star lakes are stocked with rainbow trout *Oncorhynchus mykiss*, and also have resident populations of largemouth bass *Micropterus salmoides*, bluegill sunfish *Lepomis macrochirus* and yellow perch *Perca flavescens*.

We sampled each lake 7–8 times between May 2005 and May 2006. During each sampling event, fish were collected by angling or variable mesh gill nets. Live fish were sedated on undeveloped lakes or euthanized on urban lakes using MS-222 (99.5% tricaine methanesulfonate), their gut contents were evacuated by gastric lavage, and the contents were preserved in 95% ethanol. If fish expired while in the gill nets, entire stomachs were removed in the field and preserved in 95% ethanol. Prior to gut evacuation or stomach removal, all fish were weighed to the nearest 0.1 g.

Fish diet items were identified using a dissecting microscope to a taxonomic level that allowed categorization of prey taxa into habitat of origin. Whole stomachs were dissected and the contents removed and identified using a dissecting microscope. All individual prey items in each diet were counted, and prey taxa were categorized as pelagic, benthic or terrestrial in origin. A subset of individuals ($n = 36$, mean) of each prey taxa from each lake was used to generate lake-specific length–dry mass regressions.

Prey dry mass values for calculating diet proportions were determined either by directly measuring the dry mass of prey items or, more frequently, using prey lengths in combination with length–mass regressions. A per-taxa average length was calculated for each individual diet. Mean dry mass for each prey taxa was then determined for each diet using length–mass regressions, and multiplied by the number of individuals in the diet to give the total dry mass per diet for each taxa. When developing length–mass regressions was impossible owing to a dearth of intact samples, length–mass regressions from the literature were used (Sample et al. 1993, Benke et al. 1999, Sabo et al. 2002, Chimney et al. 2007). Based on total dry mass per taxa, proportions of the diet comprised of prey from benthic, pelagic and terrestrial habitats were calculated for each fish. Whole-lake means of diet proportions were calculated across all individual fish, assuming that the relative representation of different species in gillnets reflected their relative abundance in the lake.

To estimate the relative importance of terrestrial prey items to fish, we compared consumption, daily ration, gut fullness and energy intake of trout, the fish taxa that was common to all lakes. For each fish, maximum consumption C_{max} was calculated according to the bioenergetics model for *Oncorhynchus mykiss* described by Rand et al. (1993) as

$$C_{max} = 0.628 \times W^{-0.30} \quad (1)$$

where W is fish wet mass (g). C_{max} has units of g wet prey mass per g wet fish mass per day. This value for C_{max} represents maximum consumption at 20°C (Rand et al. 1993) and is therefore a liberal estimate of maximum consumption.

Daily ration (D) was calculated according to a modified Eggers model (Elliott and Persson 1978, Eggers 1979, Principe et al. 2007) as

$$D = F \times R_e \times 24 \quad (2)$$

where F is gut fullness, and R_e is evacuation rate (h^{-1}). Daily ration has units of g prey wet mass per g fish wet mass per day. Gut fullness (F) was calculated according to Principe et al. (2007) as

$$F = \frac{G}{W} \times 100 \quad (3)$$

where G is gut content wet mass (g) and W is individual fish wet mass (g). Gut fullness is assumed to be the average gut fullness over a 24-h period. Gastric evacuation rate (R_e) was calculated following He and Wurtsbaugh (1993) as

$$R_e = 0.049e^{0.072 \times T - 0.060\log_e(PS)} \quad (4)$$

where T is temperature (°C) and PS is the mean prey wet weight (g).

We compared the allometrically-corrected feeding rate for fish in each lake according to:

$$p = \frac{D}{C_{max}}$$

where p refers to the proportion of maximum possible consumption rate (based on physiology) observed in each fish (Eby et al. 1995).

Daily energy ration (DE) was also calculated for each fish, as

$$DE = D \times J$$

where J is joules per g of prey wet weight. Prey energy densities and dry weight: wet weight conversions were collected from the literature (Cummins and Wuycheck 1971, Yako et al. 2000, Gray 2005, Koehler et al. 2006) or personal communications (Beauchamp pers. comm.).

Survey of fish diets along urbanization gradient

Between June and August 1998, 28 Pacific Northwest lakes, including the above four lakes, were sampled one time. The lakes have been described in more detail in Moore et al. (2003) and were selected to fall along a gradient of shoreline development ranging from wholly undeveloped to lakes with 100% of the shoreline developed. Development intensity was calculated as the percent of shoreline containing human residential or recreational development within 10 meters of shore. The lakes were monomictic, between 34–522 m above sea level, and had a mean depth of 5.2 m (±1.0 SE), an average surface area of 18.8 ha (±3.6 SE) and an average watershed area of 285 ha (±54 SE) (Bortleson et al. 1976). Fish were collected with gill nets and processed as above, except that on each lake, the gut contents of all similarly-sized individuals of each fish taxa were combined, and per-lake, per-fish species averages were calculated for fish mass, prey taxa frequency (i.e. number of individuals per taxa for each diet), and prey taxa length. Prey habitat of origin, dry weights, and individual and whole-lake means of diet proportions were determined as above.

Literature survey

The published literature was searched for reports of the diet composition of trout *Oncorhynchus* spp., bass and yellow perch. Only studies that assigned prey items into benthic, pelagic, and terrestrial categories were considered (Appendix 1). Many papers could not be included because benthic and terrestrial insects were lumped into a single group. Shoreline development intensity information was collected either from the published studies, from personal communications with the study authors, or estimated using Google Earth (Mountain View, California, USA). Whole-lake means of diet proportions were determined as above.

Statistical analyses

All statistical analyses were conducted using Systat 11.0. Proportional data were arcsine-square root transformed to improve normality. We tested for seasonal and development effects on the proportion of terrestrial insects in fish diets using ANOVA and Tukey's LSD post-hoc tests ($\alpha = 0.05$) with date, development and date × development as main effects. We tested for the effect of development on gut fullness, daily ration and daily energy ration using Student's t-tests ($\alpha = 0.05$).

Results

Focused comparison

Shoreline development was associated with reduced quantities of terrestrial insects in fish diets on annual and seasonal scales. Across all sampling events, 40% of fish sampled in the undeveloped lakes consumed terrestrial insects, and >25% of fish in undeveloped lakes consumed at least 10% terrestrial insects (Fig. 1). In contrast, a very small minority (2.4%) of fish sampled in developed lakes had terrestrial insects in their diets, and only in very small proportions of total diet mass (Fig. 1).

Seasonally, the proportion by mass of terrestrial insects in fish diets varied greatly on four Pacific Northwest lakes, from 0–100% (Fig. 2). The terrestrial diet proportion was significantly higher in undeveloped lakes as compared to developed lakes across all seasons ($F_{1,269} = 13.68$, $p < 0.001$), with the diets of fishes in the undeveloped lakes composed of up to 100% terrestrial insects, as compared to a maximum of 2% in the two developed lakes. These results were similar whether we included all fish taxa in the developed lake data, or whether we considered only the diets of trout. There was a significant effect of date on the proportion of terrestrial prey in diets of fish from

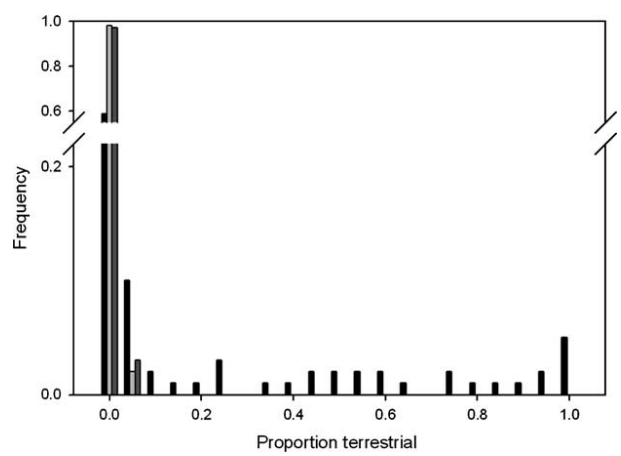

Figure 1. Frequency distribution of the proportion of individual fish diets, by dry mass, comprised of terrestrial prey in four lakes in the Pacific Northwest. Shown are diets of cutthroat trout from two undeveloped lakes (dark bars, n = 182), and rainbow trout (dark grey bars, n = 77) and all fish taxa (light grey bars, n = 125) from two developed lakes. Note break in scale on y-axis from 0.25 to 0.55.

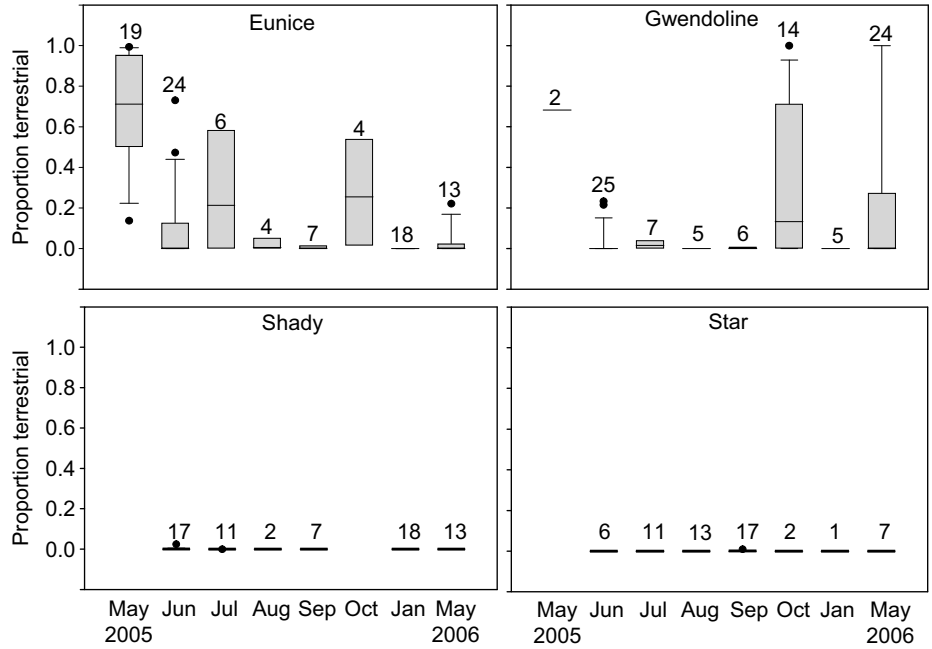

Figure 2. Seasonal patterns of proportion, by mass, of fish diets comprised of terrestrial insects from four Pacific Northwest lakes. Boxes represent medians (vertical line) plus the central 50% of all values. Symbols represent outliers, numbers above bars are sample sizes. Eunice (n = 95) and Gwendoline (n = 88) lakes are undeveloped, Shady (n = 68) and Star (n = 57) lakes are developed. Urban lakes were not sampled in May 2005.

undeveloped lakes ($F_{6,151} = 4.49$, $p < 0.001$), and post-hoc tests revealed that terrestrial diet proportions were significantly greater in October than in other months (Fig. 1). There was no significant seasonal variation in terrestrial diet proportions in fish from developed lakes ($F_{6,188} = 0.52$, $p = 0.79$). The most common prey taxa in fish diets were chironomid and other dipteran pupae, *Daphnia* spp., amphipods and isopods (Table 1). Annual whole-lake averages of the terrestrial contribution to fish diets in undeveloped lakes were 17.4% (Eunice) and 13.9% (Gwendoline) of dry mass, versus 0.0% for both developed lakes.

There was considerable seasonal variation in daily energy ration (DE) and the relative contribution of benthic, pelagic, and terrestrial prey to DE in both the undeveloped and developed lakes (Fig. 3), though daily energy ration was on average 35% greater per season in undeveloped lakes (Table 2). Benthic resources contributed the most energy to trout on both developed and undeveloped lakes, across all months. Terrestrial insect contributions to daily energy ration were biggest in undeveloped lakes in May and June 2005. Total energy intake followed predictable seasonal trends associated with water temperature, such that intake was greater in warmer months and reduced in cold, winter months (Fig. 3).

Across all sampling events, mean gut fullness was twice as large in trout from undeveloped lakes (two sample t-test, $p = 0.001$) and daily ration, which combines gut fullness with evacuation rate, was nearly twice as large in trout from undeveloped lakes (two sample t-test, $p = 0.02$) compared to the two developed lakes (Table 2). Daily energy ration was on average 50% greater in trout from undeveloped lakes (two sample t-test, $p = 0.0003$; Table 2). Daily energy ration owing to benthic prey was significantly greater on undeveloped lakes as compared to developed lakes (two sample t-test, $p = 0.001$). C_{max} was greater on undeveloped lakes (two sample t-test; $p < 0.0001$) because fish were smaller in those lakes, but daily ration as a percent of C_{max} (p) was not significantly different among lakes or between lake development types (Table 2), averaging 37.5% across all fish. Mean prey energy density (J g^{-1} prey wet mass) was greater for terrestrial insects than either benthic or pelagic prey items (Table 3).

Changes in terrestrial insect subsidies along an urbanization gradient

In the survey of 28 Pacific Northwest lakes, consumption of terrestrial prey in undeveloped lakes was often high but variable (from 0–81% of dry mass). In lakes with greater than 10% shoreline development, terrestrial contributions to fish diets were not detected. Data from the literature on North American lakes also showed the loss of terrestrial insects from fish diets on lakes with high levels (i.e. > 50%) of shoreline development (Fig. 4).

Predation on terrestrial insects by trout, bass, and yellow perch declined with shoreline development in 28 Pacific Northwest lakes and 24 North American lakes (Fig. 5). Predation on terrestrial insects varied in low development and undeveloped lakes, and terrestrial prey items were absent from diets at lakeshore development intensity above 50%. Trout consumed higher proportions of terrestrial prey than either bass or perch. In fact, terrestrial prey were not detected at all in perch diets, though only one lake with 0% development was included in the analysis of perch diets.

Table 1. The most common prey items in diets of fishes in four Pacific Northwest lakes. Values for most common taxa represent the proportion of fish diets in which each prey taxon appears. Values for most abundant taxa represent the proportion of diets in which the prey item is the most abundant by dry weight.

	Most common	Most abundant
Eunice	Diptera pupae (0.54) Daphnia (0.36) Trichoptera larvae (0.26) Spider (0.23)	Daphnia (0.29) Diptera pupae (0.18)
Gwendoline	Diptera pupae (0.60) Amphipoda (0.34) Daphnia (0.31) Ephemeroptera larvae (0.26)	Diptera pupae (0.18) Daphnia (0.17)
Shady	Diptera pupae (0.48) Chironomidae larvae (0.44) Chaoborus larvae (0.38) Daphnia (0.34)	Isopoda (0.30) Daphnia (0.19)
Star	Chironomidae larvae (0.69) Diptera pupae (0.45) Daphnia (0.41) Amphipoda (0.32)	Chironomidae larvae (0.39) Daphnia (0.23)

Discussion

Extensive research in stream ecosystems has highlighted the importance of linkages between riparian vegetation and aquatic food webs as terrestrial invertebrates are a substantial source of energy for stream fishes. This study expands this concept to lake ecosystems, demonstrating that lakeshore riparian habitat performs the same function as streamside riparian habitat, namely in supporting the delivery of terrestrial insects to surface waters where they are consumed by fishes. The riparian vegetation density on the urban lakes in the focused comparison was approximately one-tenth that of the undeveloped lakes (Francis and Schindler 2006), and the corresponding decrease in the terrestrial portion of fish diets suggests that this deforestation has direct effects on fish diets. At a broader geographical scale, a 2002 survey of 20 of the 28 of the Pacific Northwest lakes in the present study demonstrated a decrease in riparian forest density with urban shoreline development (Francis and Schindler 2006). Though not directly measured, understory riparian vegetation also decreased along the same gradient, and the results here indicate that concomitant with this riparian denudation is a decrease in terrestrial insects in fish diets. Furthermore, the intensity of lakeshore urban development appears related to the importance of terrestrial prey for fishes, as terrestrial insects were not detected in diets at urbanization levels above 50% shoreline development.

The prevalence of terrestrial insects in fish diets clearly declined with shoreline development in the 28 lake, single-sample, survey. In the four lakes, where more intensive sampling accounted for seasonality, patterns were less clear. Annual means of the proportion of terrestrial insects in fish diets from the intensive, semi-regular sampling of four lakes were qualitatively similar to the pattern observed in the survey, despite seasonal cycles that may control inputs of terrestrial insects (Nakano and Murakami 2001) and evidence that insect inputs to aquatic ecosystems are pulsed and stochastic (Carlton and Goldman 1984). Our intensive sampling of four Pacific Northwest lakes likely missed some major insect deposition events, as evidenced by the high variation in terrestrial diet proportions in undeveloped lakes (Fig. 3). Presumably terrestrial insect flux events on urban lakes were missed with roughly equal probability and therefore comparisons between lake types are relevant on both annual and finer time scales. Because of heavy exploitation in the developed lakes, we were unable to catch fish in urban lakes on all sampling dates, most notably in May and October 2005, months with substantial terrestrial insects in fish diets on undeveloped lakes. However, the absence of terrestrial insects on developed lakes was so consistent across all sampling

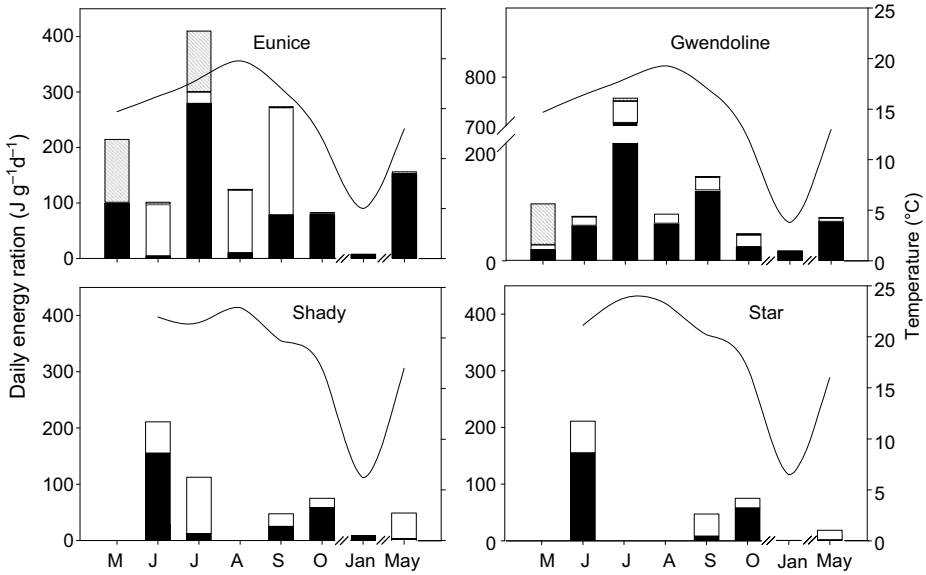

Figure 3. Daily energy rations for four lakes in the focused comparison. Different colors in bars represent different prey habitat type: dark fills are benthic, open fills are pelagic, and hatched fills are terrestrial. Note break in scale on y-axis for Gwendoline Lake. Lines represent surface temperatures for each sample period.

Table 2. Consumption and energy intake rates for trout *Oncorhynchus* spp. in four Pacific Northwest lakes. Eunice and Gwendoline lakes are undeveloped, Shady and Star lakes are developed. All values are annual means. Fish mass is in grams of wet mass; gut fullness is in grams of prey wet mass per gram of fish wet mass; daily ration is in grams of prey wet mass per gram of fish wet mass per day; C_{max} is in grams of prey wet mass per gram fish wet mass per day; p is the ratio of gut fullness to C_{max}.

Lake	Fish mass (g)	Gut fullness ($g\,g^{-1}$)	Daily ration ($g\,g^{-1}$)	C_{max} ($g\,g^{-1}\,d^{-1}$)	p
Eunice	62.1	0.005	0.025	0.154	0.147
Gwendoline	86.2	0.003	0.016	0.154	0.101
Shady	348.2	0.002	0.012	0.088	0.121
Star	292.4	0.002	0.015	0.107	0.136

periods, and in both Shady and Star lakes, it is more than likely that our sampling regime captured the pattern of terrestrial diet proportions in those lakes. Furthermore, the coherence between patterns observed in Pacific Northwest lakes and those from lakes around North America and across multiple fish taxa indicates that our intensive sampling captured the general pattern of terrestrial insects in fish diets, that riparian vegetation serves a similar function among geographic locations, and that terrestrial insects can be an important subsidy, regardless of eco-region or forest type.

Across all sampling events, gut fullness and daily ration were greater on the undeveloped lakes (Table 2). These patterns in consumption and ration are expected based on allometric relationships, which account for the fact that smaller fish have higher mass-corrected ration (Rand et al. 1993). Therefore, the smaller average size of fish in the two undeveloped lakes (Table 2) could explain the greater gut fullness and ration sizes observed in those lakes. A more informative metric of consumption rate, p, daily ration as a percent of C_{max}, was roughly equal between developed and undeveloped lakes, indicating fish in undeveloped lakes are not consuming at significantly higher rates than their counterparts in developed lakes, despite the addition of terrestrial prey. However, higher daily energy rations (Table 2) and greater mean prey energies (Table 3) indicate important and significant differences in diet quality between the two lake types that may translate into greater growth of the fish from undeveloped lakes. These results suggest that lakes with intact shorelines have greater capacity to grow and sustain fish populations, in part due to the importance of the increased inputs of terrestrial prey.

The four lakes studied varied dramatically in terms of fish exploitation and density, resulting in strong differences in mean size of fish, and especially trout species, with rainbow trout in developed lakes being much larger than the cutthroat trout in the undeveloped lakes. While we did not directly estimate fish population sizes in each lake, we observed that fish relative density was much higher in the undeveloped lakes. Because of this, we might expect that competition for prey resources among fish in the undeveloped lakes is greater than in the developed lakes. Therefore, one potential alternative explanation for these results is that consumption of terrestrial prey simply reflects prey switching to a less desirable prey source (i.e. terrestrial prey) owing to increased competition for the preferred diet items (i.e. benthic invertebrates). Furthermore, higher productivity of the urban lakes owing to increased nutrient inputs (Moore et al. 2003) might lead to the assumption that the developed lakes have more productive benthic communities and therefore higher availability of benthic prey resources. However, daily energy rations owing to benthic resources were higher in the undeveloped lakes, suggesting that the patterns of terrestrial prey contributions to diets reflect changes in terrestrial prey availability. Further, previous research on these same lakes showed higher densities of benthic invertebrates in the undeveloped lakes as compared to the urban lakes (Francis et al. 2007), suggesting that, if anything, competition for benthic prey is actually greater in the urban lakes.

Variability in the contribution of terrestrial prey to daily energy ration across lakes and across seasons (Fig. 3) suggests several potential co-occurring dynamics. The frequency distribution of the proportion of terrestrial insects in fish diets (Fig. 2) is a continuous, unimodal distribution with a long tail of extreme events that reflects pulsed resources. While inputs of terrestrial insects to streams are relatively constant during the warmer months (Nakano and Murakami 2001), no such pattern has been observed on lakes. Rather, where they have been measured, terrestrial inputs to lakes are highly variable both temporally and spatially (Norlin 1964, 1967, Cole et al. 1990) and characterized by extreme deposition events (Carlton and Goldman 1984). Therefore, terrestrial insects may represent a greater portion of ration and energy per annum than is reflected in the data here, owing to a mismatch between resource pulses and sampling events. Mehner (2005) measured terrestrial insect subsidies to bleak *Alburnus*

Table 3. Energy densities of prey by habitat type, and of diet by lake. Prey energy densities are mean joules per gram wet mass of all prey items from each habitat type found in fish diets in the four Pacific Northwest lakes, based on published energy density values. Daily energy intake is the mean, across all fish, of the diet proportion (by wet mass) represented by each prey item, multiplied by that prey's energy density ($J\,g^{-1}$ wet mass).

Habitat	Prey energy density ($J\,g^{-1}$ wet mass)	Lake	p × daily energy intake ($J\,g^{-1}$)
Benthic	7154	Eunice	854.3
Pelagic	5409	Gwendoline	784.9
Terrestrial	9561	Shady	455.6
		Star	596.0

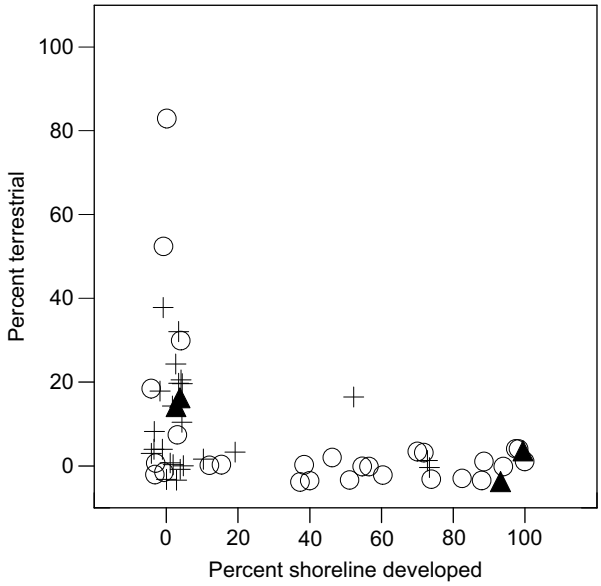

Figure 4. Percent of fish diets, by mass, of terrestrial origin from 52 North American lakes along a shoreline development gradient. Each symbol represents the annual average for a given lake/year combination, for four lakes sampled multiple times from 2005–2006 (▲), 28 Pacific Northwest lakes sampled once in 1998 (○), and 24 North American lakes from a literature review (+). Overlapping symbols are staggered for visibility.

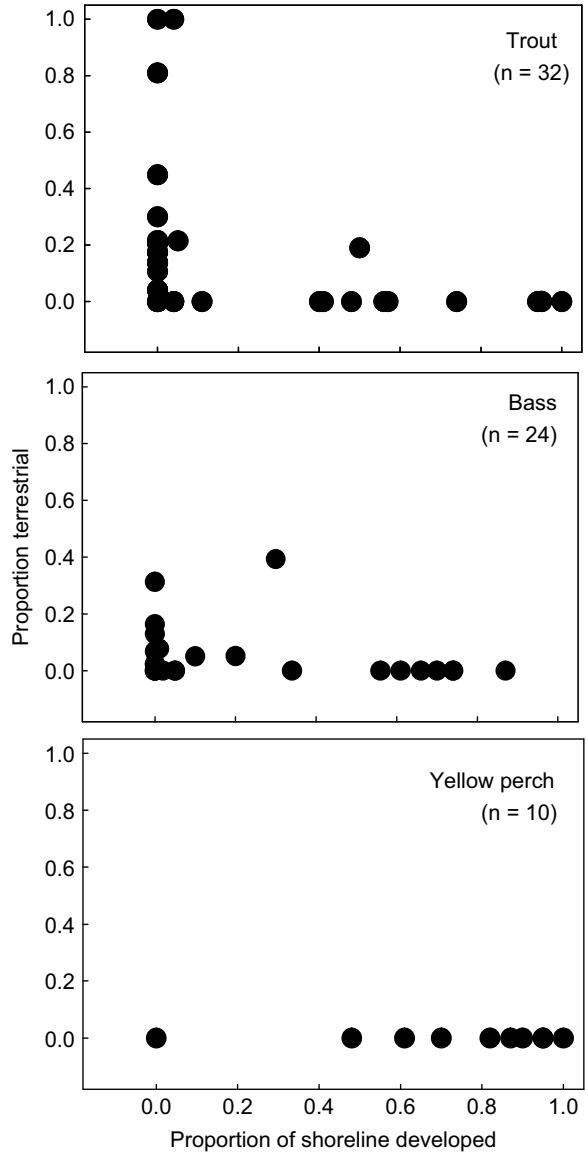

Figure 5. Proportion of diet represented by terrestrial prey in three fish taxa in 52 North American lakes along a gradient of urban development. Symbols represent means for a single lake/year pair.

alburnus, sampling monthly between April and October over two years, and found that, averaged over this time frame, terrestrial insects represented 84% of bleak diets by biomass. Here, we found that on the undeveloped lakes, terrestrial insects represented 13–17% of annual trout diets. It is possible that if terrestrial insect fluxes occur in pulses, as is suggested by the high variation observed in undeveloped lakes, repeated and more regular sampling would inflate our estimates of their occurrence in fish diets. Despite the irregular nature of the terrestrial insect subsidy in undeveloped lakes, its importance is highlighted when compared to the total absence of any terrestrial prey use by fish in urban lakes. Furthermore, given the higher energetic densities of terrestrial prey relative to aquatic taxa, the overall contribution of terrestrial subsidies towards annual growth may be greater than the proportional contribution to diets indicates.

Globally, human populations are becoming increasingly urban. The United Nations predicts that in the next 40 years, two-thirds of the world's population will live in urban areas (United Nations 2007). Pressures on ecosystems in and around urban areas are intensifying, and land-use changes associated with urbanization affect climate, biodiversity, biogeochemical cycles and hydrological cycles (Grimm et al. 2008). Because of human reliance on freshwater ecosystem services, urbanization is concentrated near coastlines (Walsh et al. 2003, Grimm et al. 2008) and therefore human effects are magnified in aquatic ecosystems. Human activities have cumulative effects on lakes, from non-point source pollution and eutrophication (Schindler 2006) to habitat alteration (Jennings et al. 2003, Francis and Schindler 2006, Francis et al. 2007). Loss of riparian habitat is an effect that is essentially unexplored, though the results from this study indicate that alteration of riparian forests likely has consequences for lake populations and food webs. Overall, these results highlight the importance of lakeshore riparian habitats as sources of terrestrial insects to surface waters and demonstrate that terrestrial insects are an important, though variable, energy subsidy for fish. How lake communities respond to reductions in these subsidies remains unknown.

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literature review. The 2005–2006 diet analyses were done by Matthew Pedersen and Anna Coogan.

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Appendix 1. References from literature search of proportional contributions of terrestrial insects to fish diets.

Lake	Location	Citation
Amchitka Island	Alaska, USA	Palmisano, J. J. and W. T. Helm. 1971. BioScience 21: 637–641
Bear Lake	Idaho/Utah, USA	Ruzycki, J. R. et al. 2001 Trans. Am. Fish. Soc. 130: 1175–1189
Beaver Creek beaver ponds	Idaho, USA	Hilderbrand, R. H. and J. L. Kershner. 2004. N. Am. J. Fish. Mgmt. 24: 33–40
Big Crooked Lake	Wisconsin, USA	Frey, A. P. and M. A. Bozeck. 2003. J. Fresh. Ecol. 18: 43–54
Buttle Lake	British Columbia, Canada	McMynn, R. G. 1953. Mgmt Publication No. 2 of the BC Game Commission.
Castle Lake	California, USA	Swift, M. C. 1970. Calif. Fish Game 56: 109–120
Castle Lake	California, USA	Wurtsbaugh, W. A. et al. 1975. Trans. Am. Fish. Soc. 1: 88–95
Deer Island Lake	Arizona, USA	Saiki, M. K. and J. C. Tash. 1978. Am. Mid. Nat. 100: 116–125
Jorgens Lake	Minnesota, USA	Soupir, C. A. et. al. 2000. Can. J. Zool. 78: 1759–1766
Lake Oahe	South Dakota, USA	Lynott, S. T. et. al. 1995. J. Fresh. Ecol. 10: 399–407
Lake Opeongo	Ontario, Canada	Martin, N. V. 1970. J. Fish. Res. Brd. Canada 27: 125–146
Lake Taneycomo	Missouri, USA	Weiland, M. A. and R. S. Hayward. 1997. Trans. Am. Fish. Soc. 126: 758–773
Lake Washington	Washington, USA	M. Mazur, pers. comm.
Lake Washington	Washington, USA	McIntyre, J. K et. al. 2006. Trans. Am. Fish. Soc. 135: 1312–1328
Little Moose Lake	New York, USA	Weidel, B. C. et. al. 2000. J. Fresh. Ecol. 15: 411–428
Locator-WarClub Lakes	Minnesota, USA	Soupir, C. A. et. al. 2000. Can. J. Zool. 78: 1759–1766
Loiten Lake	Minnesota, USA	Soupir, C. A. et. al. 2000. Can. J. Zool. 78: 1759–1766
Long Lake	Michigan, USA	Hodgson, J. R. et. al. 1997. Ecol. Fresh. Fish. 6: 144–149
Mactaquac Arm	New Brunswick, Canada	Hanson, S. D. and R. A. Curry. 2005. Trans. Am. Fish. Soc. 134: 356–368
Marion Lake	British Columbia, Canada	Efford, I. E. and K. Tsumura. 1973. Trans. Am. Fish. Soc. 1: 33–47
Oromocto Lake	New Brunswick, Canada	Hanson, S. D. and R. A. Curry. 2005. Trans. Am. Fish. Soc. 134: 356–368
Peter Lake	Michigan, USA	Hodgson, J. R. et al. 1989. Trans. Am. Fish. Soc. 118:11–19
Pomme de Terre Lake	Missouri, USA	Pope, K. L., et al. 2001. Environ. Biol. Fish. 61: 329–339
Quill Lake	Minnesota, USA	Soupir, C. A. et. al. 2000. Can. J. Zool. 78: 1759–1766
Skiatook Lake	Oklahoma, USA	Long, J. M. and W. L. Fisher. 2000. J. Fresh. Ecol. 15: 465–481
Spirit Lake	Iowa, USA	Pelham, M. E. et al. 2001. Ecol. Fresh. Fish. 10: 198–211
Stockton Lake	Missouri, USA	Pope, K. L. et al. 2001. Env. Biol. Fish. 61: 329–339

Appendix 2. Prey energy densities used in calculations of daily energy ration.

Prey taxa	Stage	J g^{-1} wet weight	Citation
Acari	Adult	21860	Cummins and Wuycheck 1971
Amphibia	Larvae	1700	Yako et al. 2000
Amphipod	Adult	2427	Gray 2005
Annelid	Adult	1981	Gray et al. unpubl.
Anisoptera	Larvae	21320	Cummins and Wuycheck 1971
Aphididae	Adult	11915	Gray et al. unpubl.
Apidae	Adult	12670	Gray et al. unpubl.
Arachnid	Adult	5321	Gray et al. unpubl.
Bivalve	Adult	22708	Cummins and Wuycheck 1971
Bosmina	Adult	3976	Koehler et al. 2006
Chaoboridae	Larvae	3976	Koehler et al. 2006
Chironomidae	Larvae	2475	Gray 2005
Chironomidae	Pupae	3400	Koehler et al. 2006
Coccinellidae	Adult	7968	Gray 2005
Coleoptera	Adult	7968	Gray 2005
Copepoda	Adult	3976	Koehler et al. 2006
Culicidae	Adult	4859	Gray 2005
Daphnia	Adult	3976	Koehler et al. 2006
Diptera	Adult	8923	Gray et al. unpubl.
Diptera	Larvae	2581	Gray et al. unpubl.
Diptera	Pupae	3400	Koehler et al. 2006
Ephemeroptera	Adult	20429	Cummins and Wuycheck 1971
Ephemeroptera	Larvae	22872	Cummins and Wuycheck 1971
Formicidae	Adult	11125	Gray et al. unpubl.
Gastropoda	Adult	6570	Cummins and Wuycheck
Hemiptera	Adult	10927	Gray et al. unpubl.
Holopedium	Adult	3976	Koehler et al. 2006
Hymenoptera	Adult	12230	Gray 2005
Isopoda	Adult	3335	Gray 2005
Isoptera	Adult	12230	Gray et al. unpubl.
Lepidoptera	Larvae	8502	Gray et al. unpubl.
Notonectidae	Adult	20170	Cummins and Wuycheck 1971
Oligochaete	Adult	1981	Gray 2005
Pipunculidae	Adult	8923	Gray et al. unpubl.
Polyphaemus	Adult	3976	Koehler et al. 2006
Sciaridae	Adult	8923	Gray 2005
Sialidae	Larvae	21788	Cummins and Wuycheck 1971
Staphylinidae	Adult	1866	Gray et al. unpubl.
Tenthredinidae	Adult	8923	Gray et al. unpubl.
Trichoptera	Pupae	5814	Gray et al. unpubl.
Trichoptera	Adult	7756	Gray 2005
Trichoptera	Larvae	5814	Gray et al. unpubl.
Zygoptera	Adult	23674	Cummins and Wuycheck 1971
Zygoptera	Larvae	21742	Cummins and Wuycheck 1971

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Effects of Urbanization on the Dynamics of Organic Sediments in Temperate Lakes

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Abstract

Residential development of lakeshores affects the structure and function of riparian and littoral habitats. Organic detritus in sediments is a critical component of littoral food webs, but the effects of urbanization on sediment characteristics are unexplored. We characterized the quantity of organic sediments in Pacific Northwest lakes along a development gradient and found a 10-fold decline in the proportion of detritus in littoral sediments associated with density of lakeshore dwellings. In a comparison between two fully developed lakes and two undeveloped reference lakes, we examined several possible controls on sedimentary organic content, including terrestrial inputs, decomposition rates and associated macroinvertebrate communities, and physical retention by coarse wood. The littoral sediments of undeveloped lakes ranged from 34 to 77% organic by mass, whereas the range on urban lakes was an order of magnitude less, ranging from 1 to 3% organic. We found no significant differences in terrestrial litter inputs between the two sets of lakes. Leaf litter decomposition rates did not vary significantly between the two sets of lakes, and we found higher densities of shredder macroinvertebrate taxa in the littoral zones of undeveloped lakes. Sedimentary organic matter on undeveloped lakes accumulated in shallow waters and declined with distance from shore, whereas the opposite pattern existed on urban lakes. Our results suggest that coarse wood physically retains organic matter in littoral zones where it can enter the detrital energy pathway, and the loss of this feature on urban lakes alters littoral sediment characteristics, with potentially far-reaching consequences for lake food webs.

Key words: littoral; urban ecology; coarse woody debris; sediments; organic matter; leaf litter; aquatic-terrestrial coupling; benthic invertebrates.

Introduction

As urban development sprawls, the effects of human activities on lake ecosystems are becoming more apparent and complex. For several decades, the greatest human threat to lake ecosystems was eutrophication of surface waters (Schindler 1978; Carpenter and others 1998). Though eutrophication owing to nonpoint source pollution from agricultural, urban, and industrial sources remains a problem for lakes in North America (Carpenter and others 1998; Moore and others 2003), it is apparent that human activities have myriad effects

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on lake ecosystems. For example, fish growth rates are negatively correlated with lakeshore development intensity (Schindler and others 2000), and the spatial distribution of fishes is altered by urbanization (Lewin and others 2004; Scheuerell and Schindler 2004). Lakeshore and watershed urbanization are correlated with reduced amphibian abundance (Woodford and Meyer 2003) and shifts in benthic invertebrate (Quinlan and others 2002) and zooplankton (Dodson and others 2005) community composition.

Human activities and their impacts on limnetic ecosystems are concentrated along lakeshores, where coupling between terrestrial and aquatic systems is particularly strong. Residential development of shorelines is associated with reductions in riparian forest density as homes, lawns, and related residential structures replace native vegetation (Christensen and others 1996; Francis and Schindler 2006; Marburg and others 2006) and alter plant community diversity, reducing the abundance of native plant species (Elias and Meyer 2003). Humans alter littoral habitats through residential development, for example by removing macrophytes (Jennings and others 2003), which in turn changes the spatial distribution of fishes (Bryan and Scarnecchia 1992).

One unexplored consequence of lakeshore urbanization is its impact on the distribution, composition, and abundance of sedimentary organic matter. The role of detritus and organic matter as a critical energy link in lakes has long been recognized (Lindeman 1942). Recently, it is argued that detritus forms the foundation for energy flow in many aquatic ecosystems (Polis and Strong 1996; Moore and others 2004). Organic matter provides a substrate for colonization by bacteria and other microbes, which are a food source for macroinvertebrates and fishes. Because benthic energy pathways are critical to upper trophic levels in lakes (Schindler and Scheuerell 2002; Vander Zanden and Vadeboncoeur 2002), the loss of this detrital energy source could have major consequences for food web structure and lake ecosystem function.

Here, we hypothesize that the loss of coarse wood from urban littoral habitats increases the flow of organic sediments to deeper waters, resulting in lower organic content in littoral sediments. One alternative hypothesis is that decreased inputs of terrestrial leaf litter from thinned urban riparian forests reduce littoral detritus, as development intensity is also associated with riparian deforestation on this same set of lakes (Francis and Schindler 2006). Leaf litter inputs to lake surface waters have seldom been quantified, and yet they can be substantial (Gasith and Hasler 1976; France and Peters 1995). Because allochthonous sources can dominate the organic pools in littoral sediments (Pieczynska 1990a), it is possible that the losses of particulate terrestrial inputs associated with riparian deforestation may be implicated in reduced littoral detritus pools. Another potential explanation for the lower organic content of urban lake sediments is that developed lakes have higher rates of decomposition, and therefore particulate organic matter is more rapidly processed and transferred into either the invertebrate pool or the dissolved organic matter pool. Microbial degradation can increase under higher nutrient conditions (Oertli 1993; Bayo and others 2005), and certain detritivorous macroinvertebrates thrive under eutrophic conditions associated with urbanization (Kashian and Burton 2000). Higher rates of invertebrate- or bacteria-driven decomposition in urban lakes may therefore explain losses of littoral sediment organic matter along a gradient of urbanization.

Loss of coarse wood (dead and downed tree stems and branches, here defined as >1 m in length and >10 cm in diameter) from littoral habitats is strongly associated with shoreline urbanization (Christensen and others 1996; Francis and Schindler 2006; Marburg and others 2006). Although it is assumed that wood is important in the structure and function of littoral habitats (Schindler and Scheuerell 2002; Jennings and others 2003), as yet very little is known about the specific roles played by coarse wood in lakes. It has been shown extensively in streams that coarse wood increases organic matter retention (Bilby 1981; Bilby and Ward 1991), and less coarse wood in urban lakes may result in a loss of sediment organic matter. The majority of detritus in lakes is deposited in shallow water zones (Piezynska 1990b) where it then becomes part of the sediments. In the absence of coarse wood as a physical structure to retain particulate detritus in shallow waters, organic matter deposited in littoral zones may be transported to deeper waters, resulting in reduced organic content of urban littoral sediments. This reduction in littoral sediment quality may in turn have consequences for upper trophic levels and lake-wide food web interactions, as littoral habitats house a variety of benthic invertebrates that serve as prey items for fishes and are therefore key sites for benthic–pelagic coupling (Schindler and Scheuerell 2002; Vander Zanden and Vadeboncoeur 2002).

In this paper, we investigate the effects of shoreline urbanization on littoral habitat characteristics, specifically the organic content of littoral sediments. We then explore several possible mechanisms to

explain the relationship between lakeshore development and sediment organic matter. Previous results showed a relationship between urbanization and loss of littoral coarse wood (Francis and Schindler 2006), therefore we mapped the distribution of sediments with distance from shore, hypothesizing that littoral coarse wood retains organic matter in littoral sediments. Because the riparian zones of urban lakes in this region are deforested (Francis and Schindler 2006), we explored whether reduction in leaf litter inputs results in lower organic matter in urban sediments compared to undeveloped lakes with intact riparian forests. Finally, because decomposition processes can control sediment characteristics, we measured decomposition rates on urban lakes and compared them to those observed on undeveloped lakes to determine whether biodegradation explains sediment organic content.

Methods

We sampled 15 lakes in western Washington State and southern British Columbia, Canada (see Francis and Schindler 2006 for physical descriptions). The lakes are all located in the western hemlock (*Tsuga heterophylla*) zone of the Cascade Range and Puget Trough regions. Study lakes span a gradient of urban development and are mostly located in the urban fringe of the Seattle Metropolitan area. Nearly all lakes were closed to boats with outboard motors. Three undeveloped lakes were selected as reference systems, located in the University of British Columbia's Malcolm Knapp Research Forest. The latitudinal gradient between our study sites is not sufficient to affect forest or invertebrate community composition.

We surveyed the lakes between July and October of 2002 and in October 2006 for development intensity, coarse wood density and basal area, and sediment composition. A full description of the survey and coarse wood sampling strategy is given in Francis and Schindler (2006). On each lake, we selected 4–8 transects measuring 30 m along the lakeshore, ensuring to select an even number of both leeward and windward sites. Along each transect, we enumerated and measured the diameter of all pieces of coarse wood (≥10 cm in diameter, ≥1 m in length) within or intersecting the 0.5 m depth contour. We calculated coarse wood basal area for each plot as the sum of all individual log basal areas, defined as

$$\text{basal area} = \pi * r^2 \quad (1)$$

where r = radius at breast height (trees), or where the log intersected the 0.5 m depth contour. We characterized sediment composition by collecting the top 5 cm of sediments in 0.5 m of water using a 10 cm diameter hand-held sediment corer. Sediments were refrigerated to limit further biotic processing until they were dried at 60°C to a constant mass and then combusted at 550°C to determine ash free dry mass (AFDM). Sediment organic matter was calculated as the proportion of dry mass lost during combustion.

To investigate the mechanisms driving the relationship between lakeshore urbanization and sediment organic matter observed in the survey, we conducted a focused comparative study between two urban, fully developed lakes (that is, >95% of shoreline developed; Star and Shady Lakes) located in suburban areas of Seattle and two wholly undeveloped lakes (Loon and Gwendoline Lakes) in British Columbia (Figure 1). These two sets of lakes represent the high and low ends of the gradients for sediment organic matter and residential development observed in the survey (Francis and Schindler 2006; Table 1).

We measured leaf litter input on the set of four lakes by establishing three transects running perpendicular to shore at randomly selected sites along the shoreline. Along each transect, we measured the lateral distribution of terrestrial leaf litter inputs to surface waters from July 2003 to April 2004 by deploying litter traps, 20 cm × 30 cm (0.06 m^2) floating plastic bins, at 1, 5, 10, 20, and 40 m from shore. The litter traps passively sampled aerial litterfall constantly for 10 months. Contents were removed every 22–61 (median: 46) days and dried at 60°C to a constant mass. In September 2003, a heavy rainstorm raised the water level of Loon Lake such that all traps were swamped. In February 2004, Gwendoline Lake was frozen and traps were inaccessible; however, bins continued to collect litter inputs, and therefore the sample collected in March was assumed to include inputs from the previous period.

To measure decomposition rates in the four lakes, we incubated red alder (*Alnus rubra*) leaves in mesh bags anchored to the lake bottom at two depths during late summer/autumn and winter. We collected senescent alder leaves from the watershed surrounding each lake, dried them at room temperature, and created leaf packets which were placed in either fine (<0.5 mm) or coarse (1 cm) mesh bags. The fine mesh bags successfully prevented the colonization of leaf packets by macroinvertebrates. Bags were weighed before incubation and anchored to cinder blocks on the lake bottom at two depths, above and below the thermocline, to incorporate potential variation in

Figure 1. Shorelines of Pacific Northwest lakes of different residential development intensity. **A** Gwendoline Lake, British Columbia, undeveloped; **B** Shady Lake, Washington State, 100% of shoreline developed.

Table 1. Lake Biophysical Characteristics

Lake	Location	Maximum depth (m)	Mean depth (m)	Surface area (ha)	Residential density (houses m^{-1} shoreline)	Coarse wood density (pieces m^{-1} shoreline)	Sediment organic matter (proportion by mass)	Epilimnetic Chlorophyll-a (μl^{-1})
Gwendoline	19°49′N 34°122′W	27	13.4	13.0	0	425.0	0.77	0.69
Loon	19°49′N 34°122′W	55	26	48.6	0	516.7	0.34	0.66
Shady	25°47′N 6°122′W	12	6	8.5	40	10.0	0.01	4.83
Star	21°47′N 17°122′W	15	8	14.2	40.6	0	0.03	1.32

invertebrate and microbial decomposition rates associated with temperature. Several reference bags were put through the weighing, bagging, and placement process to establish mass lost during transport, and this correction factor (−1.6% for fine bags, −6.9% for coarse mesh) was applied to all bags before statistical analyses. After the incubation period, bags were retrieved by SCUBA, dried at 60°C to a constant weight, and reweighed. Decomposition rate k (Petersen and Cummins 1974) was calculated according to an exponential function as follows:

$$W_t = W_0 e^{(-kt)} \quad (2)$$

where W_t is equal to the mass of leaf material remaining after incubating for t days and W_0 is the initial corrected mass.

We collected surface sediments at various distances from shore on all four lakes to assess the spatial distribution and particle size composition of sediment organic matter. We stratified distances into ''near'' (1–10 m) and ''far'' (20–40 m) from shore, based on previous work showing that the majority of allochthonous inputs are deposited within 10 m from shore (France and Peters 1995). We did not measure sediment organic matter in the deep basins of these lakes, but rather concentrated on the littoral zones, where organic matter accumulation was observed in the survey. We collected sediments using an Ekman dredge below each floating litter trap in summer and winter. Each sample was sorted using soil sieves into smaller than 0.06, 0.06, 0.12, 0.18, 0.42, 1.0, 2.5, and 6.35 mm size categories and dried at 60°C to a constant mass. A subsample of each was combusted and the proportion of organic content was calculated for each size category as described above.

We sampled the benthic macroinvertebrate communities in each of the four lakes along the established transects using an Ekman dredge during the late summer of 2003. We collected invertebrates in surface sediments below each floating litter trap at depths of 0.3–11 m of water. Invertebrates were preserved in 95% ethanol and identified to family.

Statistical Analyses

We used SYSTAT 11.0 (Systat Software Inc., 2004) for all statistical analyses. Data transformations—arcsine square root transformations for the proportion organic matter in sediments and natural log transformation for density of lakeshore residences—were performed prior to statistical analyses to normalize data. We used least-squares regression using data from the 15-lake survey (excluding one outlier; Studentized residual = −3.1) to assess the relationships between lakeshore residential density and sediment organic matter, and coarse wood density and sediment organic matter content. Previous work has demonstrated a relationship between lake morphometry and sediment organic matter (Rowan and others

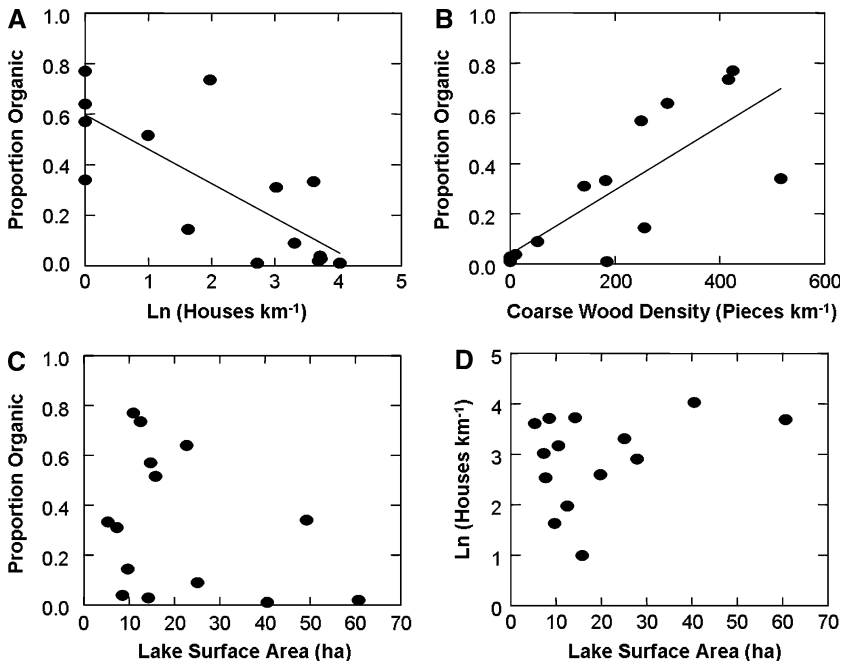

Figure 2. Organic proportion of littoral sediments as a function of **A** residential development and **B** coarse wood density; and sediment organic proportion **C**, and density of shoreline residences **D** as a function of lake surface area. All data shown are from a survey of 15 Pacific Northwest lakes, excluding one outlier (n = 14). Proportion data were arcsine-square root transformed prior to statistical analysis. Each point represents a whole lake mean.

1992), therefore we also tested for a relationship between lake size (surface area and perimeter) and sediment organic matter in the 15 lakes using least-squares regression. A final regression investigated the relationship between lake size (surface area) and residential density. In the focused comparison between urban and undeveloped lakes, we tested whether decomposition rates varied according to lake development level using ANOVA with daily mass loss rate as the response variable and development level (developed and undeveloped), season (summer and winter), depth (above and below the thermocline), and mesh size (coarse and fine) as main factors, using post-hoc tests where we found significance. To test the effect of coarse wood on organic sediment particle distribution, we used ANOVA with coarse wood density (high, low) and distance from shore as main effects on organic content of each particle size category, and we ran Student's t-tests to test for development differences within seasons at specific distances. We used a MANOVA to test for significant differences in invertebrate communities between the undeveloped and urban lakes.

Results

Sediment organic matter decreased significantly as shoreline residential development density increased in the survey of 15 lakes (adjusted r^2 = 0.60, n = 15, P = 0.001; Figure 2A). The sediment organic content varied from roughly 74% on undeveloped lakes to less than 2% on fully developed lakes. The organic matter content of littoral sediments was significantly and positively associated with the density of coarse wood (adjusted r^2 = 0.63, n = 14, P = 0.001; Figure 2B) and coarse wood basal area (adjusted r^2 = 0.68, n = 14, P < 0.001; not shown). We found no significant relationship between sediment organic matter and either metric of lake size, surface area (adjusted r^2 = 0.07, n = 14, P = 0.18; Figure 2C), or shoreline length (adjusted r^2 < 0.001, n = 14, P = 0.99), and we found no significant relationship between residential density and lake surface area (adjusted r^2 < 0.001, n = 14, P = 0.78; Figure 2D) or lake perimeter (adjusted r^2 = 0.07, n = 14, P = 0.15; not shown).

On each of the four lakes sampled for the focused comparison, annual terrestrial leaf litter inputs decreased with distance from shore (Figure 3A). We found no significant effect of development ($F_{1,17}$ = 2.5, P = 0.13) on annual leaf litter inputs to surface waters across all distances combined, though the trend was for higher inputs to urban lakes than to undeveloped lakes (Figure 3A; Table 2). Litter inputs varied with distance on all lakes ($F_{1,17}$ = 5.1, P = 0.04), decreasing with distance from shore according to a negative exponential model (developed lakes: Inputs = $158.98e^{-0.31 \times \text{Distance}}$, r^2 = 0.98; undeveloped lakes: Inputs = $66.9e^{-0.67 \times \text{Distance}}$, r^2 = 0.99). The majority (that is, >70%) of leaf litter was deposited within 5 m of shore, and

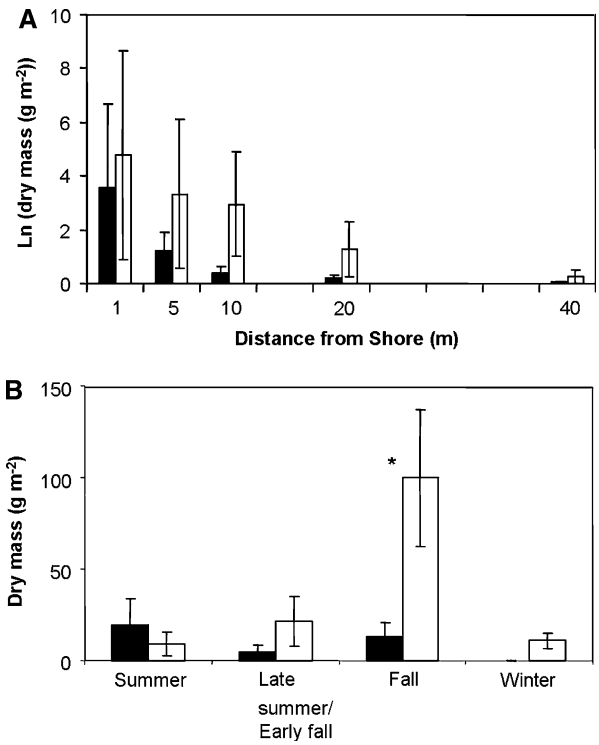

Figure 3. Terrestrial leaf litter inputs to nearshore surface waters on two undeveloped (*solid bars*) and two developed (*open bars*) lakes. **A** Annual inputs (±1 SE) at varying distances from shore. **B** Seasonal inputs (±1 SE) across all distances.

significantly less was observed beyond 10 m ($P = 0.04$). On a seasonal basis, leaf litter inputs varied significantly by development only during fall ($P = 0.04$; Figure 3B and Table 2), with higher inputs to developed lakes during that season. Peak seasonal leaf litter inputs occurred during fall on developed lakes and during summer on undeveloped lakes.

We found significant effects of season and mesh size, but no significant effect of lake development or depth relative to the thermocline, on decomposition rates across all litterbag treatments (Table 3; Figure 4). Decomposition rates were greater in summer than in winter (Tukey's HSD adjusted $r^2 = 0.22$, $df = 1$, $P = 0.006$), and were greater in coarse mesh bags than in fine (Tukey's HSD adjusted $r^2 = 0.53$, $df = 1$, $P < 0.001$), across both development types and incubation positions.

In undeveloped lakes, sediment organic matter accumulated inshore, whereas in developed lakes, sediment detritus increased with distance away from shore (Figure 5). We found significant interactions between development and distance from shore ($F_{4,19} = 22.4$, $P < 0.0001$) and distance and season ($F_{4,19} = 4.1$, $P = 0.02$), and we found a significant three-way interaction effect between development, season, and distance from shore ($F_{4,19} = 5.3$, $P = 0.005$) on the organic proportion of sediments between 1 and 40 m from the shore, such that organic matter decreased with distance from shore on undeveloped lakes but increased with distance on developed lakes. These patterns generally held in both summer and winter, except that overall organic content was higher in urban lakes in winter and more variable closest to shore in undeveloped lakes in summer. We also found significant effects of development ($F_{1,19} = 86.0$, $P < 0.0001$) and season ($F_{1,19} = 8.7$, $P = 0.008$) on the proportion organic in sediments. The mean (SE) proportion by mass of organic littoral sediments on undeveloped lakes was 0.71 (0.11), as compared to 0.11 (0.05) on developed lakes. In both seasons, the amount of sediment detritus on the two lake types converged at 40 m from shore.

Urbanization affected the size distribution of sediment particles. In particular, the smallest particles (<0.06 mm) accumulated nearshore on undeveloped lakes in greater proportions than on developed lakes (Figure 6). In fact, there were almost no fine particles in the nearshore sediments of developed lakes. In tests on the distribution of the smallest (<0.06 mm) organic sediment particles, there were significant interaction effects between development and season ($F_{4,20} = 7.3$, $P = 0.01$) and distance and season ($F_{4,20} = 4.2$, $P = 0.01$), as well as a significant distance effect ($F_{4,20} = 23.8$, $P < 0.0001$). In summer, a significantly greater proportion of fine organic particles accumulated within 5 m of shore on undeveloped lakes versus developed lakes ($P = 0.02$). In contrast, there was no significant difference in the proportion of fine organic particles in littoral sediments between developed and undeveloped lakes in winter ($P = 0.85$).

The benthic macroinvertebrate community varied substantially between urban and undeveloped lakes (Figure 7). The macroinvertebrate community composition in developed and undeveloped lakes were significantly different (MANOVA, Wilks' Lambda, $F_{7,45} = 2.18$, $P = 0.05$); specifically, we found higher densities of shredding caddisflies (Trichoptera) on the undeveloped lakes ($P = 0.02$) and higher densities of detritivorous isopods (Isopoda) on urban lakes ($P = 0.03$; Figure 7). The densities of grazing mayflies (Ephemeroptera) and predatory odonates (Odonata) were 12.9 and 10.7 m^{-2}, respectively, in the littoral zones of

Table 2. Leaf Litter Inputs to Surface Waters of Two Developed (Star, Shady) and Two Undeveloped (Gwendoline, Loon) Lakes

	Annual inputs (g dry weight)		Seasonal inputs (g dry weight)			
	Mean	SE	Summer	Late summer/early fall	Fall	Winter/early spring
Developed	165.1	63.7	8.9	21.4	99.7[a]	10.8
Undeveloped	37.4	21.2	19.6	4.3	13.1[b]	0.0

Different letters in each column indicate significant differences among lake types (Student's t-test; P < 0.05).

Table 3. Summary of ANOVA Results showing Effects of Development (developed, undeveloped), Depth (above, below the thermocline), Season (summer, winter) and Bag Mesh (coarse, fine) on Leaf Litter Decomposition Rates

Factor	Least squares mean	*F*-ratio	*df*	*P*-value
Development		0.2	1	0.70
Undeveloped	0.004			
Developed	0.004			
Depth		3.3	1	0.09
Above	0.0044			
Below	0.0035			
Season		23.5	1	0.0002
Summer	0.005			
Winter	0.003			
Mesh		55.9	1	<0.0001
Fine	0.002			
Coarse	0.006			

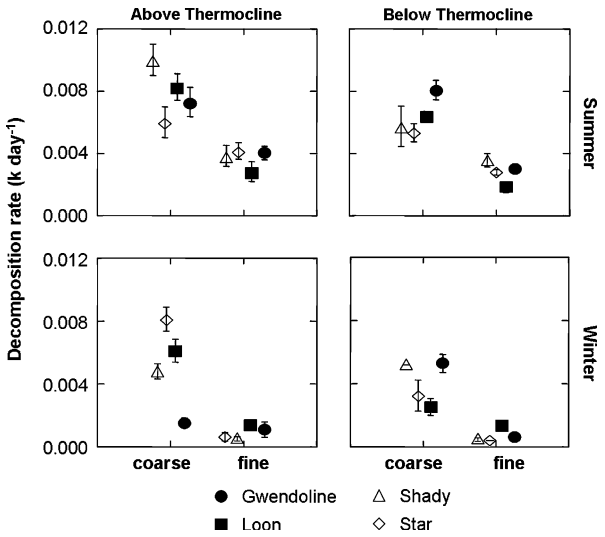

Figure 4. Daily decomposition rates of leaves incubated in two undeveloped (*filled boxes*) and two developed (*open boxes*) lakes in coarse and fine mesh bags during summer and winter. *Boxes* represent whole lake means.

undeveloped lakes, whereas these taxa were entirely absent from similar habitats on urban lakes.

Discussion

Shoreline urbanization has a variety of impacts on lake ecosystems, including eutrophication and altered littoral habitat structure, vegetation community composition, and fish growth and behavior (Schindler and others 2000; Scheuerell and Schindler 2004; Marburg and others 2006). Our results demonstrate two additional consequences of shoreline urbanization: changes in the composition and distribution of organic sediments and shifts in the benthic macroinvertebrate community. We suggest that the best explanation for depleted organic matter in urban littoral habitats is the absence of coarse wood, which provides a physical structure that retains detritus in littoral zones of undeveloped lakes. Without coarse wood to retain organic matter in shallow waters, organic matter is transported by gravity and hydrodynamics to deeper waters on urban lakes.

Riparian Litterfall Influence on Sediment Organic Matter

Our finding that leaf inputs on all lakes decrease with distance from shore according to a negative exponential model concurs with previous studies (Szczepanski 1965; Rau 1976); however, we did not find reduced annual litter inputs to developed lakes. Based on other findings that terrestrial inputs can be tightly linked to littoral organic matter (Efford 1969) and the extensive literature on the role of litterfall in lotic ecosystems, we expected that litterfall into lakes would be reduced in urban environments, where dramatic riparian deforestation has occurred (Francis and Schindler 2006). Indeed, the rare quantification of litterfall to the surface waters of lakes has been expressed in terms of litter inputs per

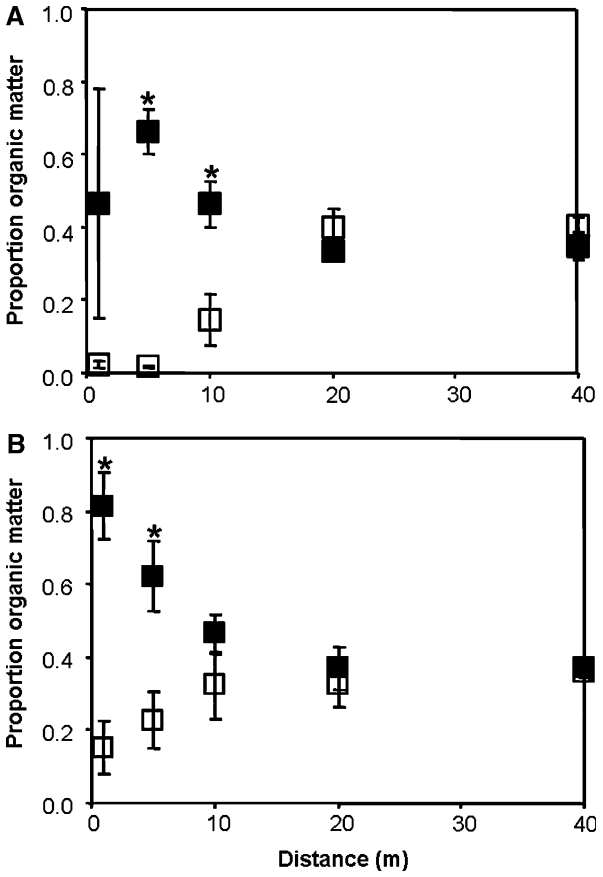

Figure 5. Proportion organic matter in surface sediments with distance from shore on two undeveloped (*filled symbols*) and two developed (*open symbols*) lakes in summer (**A**) and winter (**B**). *Error bars* are 1 SE. *Asterisks* indicate statistically significant differences (Student's *t*-test, $P < 0.05$).

meter of forested shoreline (Szczepanski 1965; Jordan and Likens 1975; Gasith and Hasler 1976).

One explanation for the absence of significantly different litterfall rates between developed and undeveloped lakes is the riparian forest composition shift that occurs with urbanization on lakes in this region. In lowland areas of the Pacific Northwest, the native riparian forest is dominated by conifers, primarily Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*; Franklin and Dyrness 1973). On the urban lakes in this study, in contrast, native forests are most often replaced with non-native, deciduous species (T. Francis, personal observation), which have considerably higher litterfall rates than conifers (Binkley and others 1992). In regions where native forests are dominated by deciduous tree species, deforestation associated with urbanization may reduce organic inputs to littoral zones of urban lakes, resulting in even greater declines in littoral detritus.

Decomposition of Sediment Organic Matter

We observed no consistent differences in decomposition rates between urban and undeveloped lakes that might explain either higher total OM or greater proportion of fine particulate OM (FPOM) in undeveloped lakes. Biological reduction of the coarse particulate OM (CPOM) pool occurs through processing and ingestion by macroinvertebrates and via microbial degradation. Both processes reduce CPOM while increasing the FPOM pool (Saunders and others 1980). Despite higher densities in the undeveloped lakes of shredding caddisflies, which are more effective decomposers relative to other shredder taxa (Bjelke and Herrmann 2005), decomposition in the presences of macroinvertebrates was not significantly higher on undeveloped lakes compared to urban lakes in any of the treatment regimes. In addition, the trend towards higher densities on undeveloped lakes of detritivorous amphipods, which ingest FPOM, could only potentially decrease the fine organic pool in these systems.

We found no difference in decomposition between urban and undeveloped lakes despite variation in some abiotic characteristics of the two sets of lakes. Beyond biotic activities, decomposition rates vary according to temperature, oxygen concentration, nutrient levels, substrate, particle size, turbulence, and pH. Our study lakes are similar in terms of annual temperature regimes, though the undeveloped lakes have a slightly higher tendency to freeze for short periods of time in winter. The urban lakes are mesotrophic (total epilimnetic P = 9–10 $\mu g\ l^{-1}$), whereas the undeveloped lakes are oligotrophic (epilimnetic P = 3–4 $\mu g\ l^{-1}$; Moore and others 2003). Despite elevated nutrient levels, however, we observed no greater overall decomposition rates in the urban lakes.

Decomposition of particulate organic matter varies across substrate type. We did not directly measure the source contributions of littoral detritus in each lake, and the trophic status of the urban lakes suggests there might be a greater proportion of autochthonous POM, which is more labile and could be processed more rapidly (Hicks and others 1994), reducing the total OM pool. Given the equivalent leaf litter inputs in both lake types, however, different detritus composition does not explain the pattern of organic matter we observed. Decomposition rates of terrestrial leaf litter also vary by litter type. For example, deciduous alder leaves decompose more quickly than more recalcitrant conifer needles (Webster and Benfield

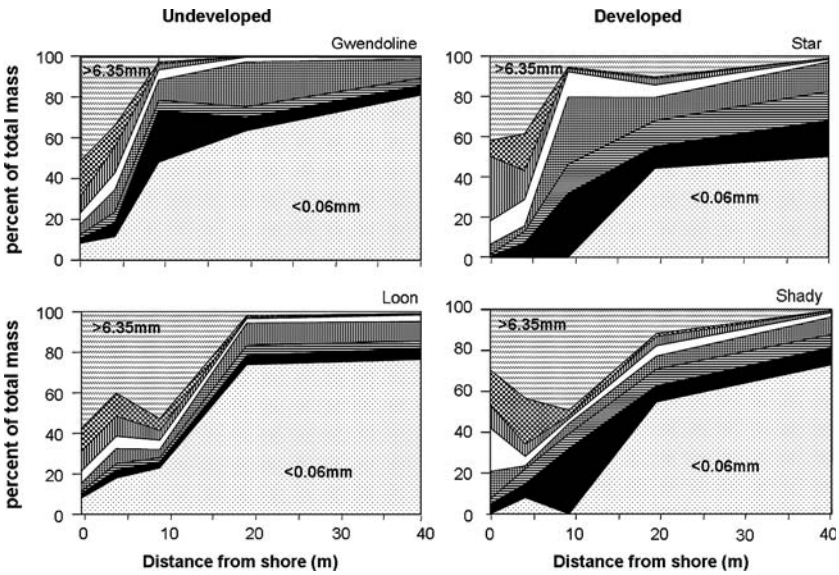

Figure 6. Size composition of organic sediment particles with distance away from shore on four Pacific Northwest lakes in summer. Particles are separated into size categories: smaller than 0.06, 0.06, 0.12, 0.18, 0.42, 1.0, 2.5 and 6.35 mm. Distributions shown are organic proportions by mass of surface sediments.

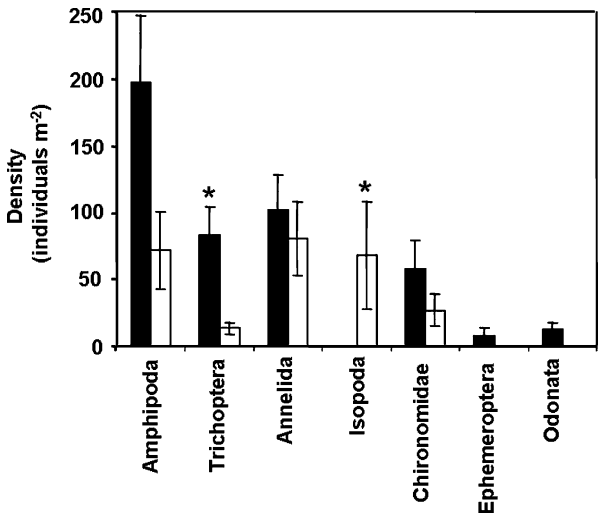

Figure 7. Macroinvertebrate densities on two undeveloped (*solid bars*) and two developed (*open bars*) lakes. *Asterisks* indicate statistically significant differences (Student's *t*-test, $P < 0.05$), *error bars* are standard errors.

1986). The riparian forest community on the undeveloped lakes is dominated by conifers, versus deciduous trees on urban lakes. Thus, it may be that urban litter is decomposed more quickly, leaving less total organic matter in shallow water sediments. Increased litter processing, however, would result in an accumulation of FPOM in nearshore sediments on urban lakes, which we did not observe.

We observed striking differences in the macroinvertebrate community composition between urban and undeveloped lakes that may be associated with reductions in coarse wood and organic sediments. In addition to the differences stated above in the shredder taxa, we found a complete loss of predatory odonates (Odonata) and grazing mayflies (Ephemeroptera) from the urban lakes. The absence of these taxa may be associated with a suite of alterations to urban littoral habitats, including the loss of coarse wood, which serves as habitat, food resource, and predation refuge. These larger invertebrate taxa are key prey for native trout, and their absence may portend a decline in the ability of urban lakes to support productive native fish populations (Schindler and others 2000).

Organic Matter Distribution and Retention

Overall, the amount of organic material in littoral sediments in undeveloped lakes was significantly greater than in urban lakes. Sediment organic matter accumulated in shallow waters on undeveloped lakes and decreased with distance from the shore, but on urban lakes, organic matter increased with distance from the shore. In the 15-lake survey, we found a significant positive correlation between the proportion of organic matter in littoral sediments and coarse wood density, and in the focused comparison, coarse wood density was significantly greater on the two undeveloped lakes compared to the urban lakes. Based on these data and the organic particle distribution, we suggest that littoral OM accumulation results from physical retention by coarse wood that impedes sediment focusing and retains POM in shallow waters. This build-up of organic matter around wood has been extensively observed in lotic ecosystems, where

Attachment 2

large wood adds hydraulic complexity to rivers and streams (Bilby and Ward 1991; Bisson and Bilby 1998) and retains both coarse and fine organic matter (Bilby 1981; Harmon and others 1986) where it is subjected to biodegradation processes.

The generality of the relationship we have found here between woody debris and sediment organic matter relies on, among other things, inputs of woody debris and production of organic matter. Accumulation of sediment organic matter in littoral zones is regularly observed in shoreline areas with submerged vegetation (James and Barko 1990; Wetzel 1990). Littoral-wetland zones are areas of high organic matter loading and degradation (Pieczynska 1990a; Wetzel 1990; Gude and others 2004); hence, accumulations at lake margins in these systems are common despite the absence of coarse wood. Lakes with significant portions of riparian habitat in wetlands, desert, agriculture, tundra, or otherwise lacking forest would not receive the inputs of woody debris found in these lakes and would not fit the pattern we have observed. Likewise, lakes with unforested watersheds, such as alpine systems, may receive lower inputs of particulate OM and not accumulate littoral detritus to the same degree as the lakes sampled here. Although the evidence is limited, existing research shows that sediment OM is not a function of lake trophic status (Rowan and others 1992). Instead, sediment OM in lakes has been correlated with maximum lake depth, lake surface area and catchment area: lake area (Rowan and others 1992), although we did not find such relationships here.

Human Behavior and Littoral Sediments

We did not directly monitor human behavior, so we cannot exclude the potential that the alteration of shorelines and littoral zones by urban lake residents contributes to the sediment patterns we observed. Certainly human behavior affects the density of coarse wood on urban lakes, as people clear littoral zones for swimming and other recreational activities. Furthermore, residents often deposit gravel or other non-organic substrates along their shoreline, creating a homogenous, non-organic littoral zone (T. Francis, personal observation). Our sampling, however, captured a variety of residential littoral zone types, including those with gravel and those without. Another human influence on sediment composition is erosion from lakeshore development which likely mobilizes soils into lakes. When these lakes were initially deforested at the turn of the twentieth century, as well as during subsequent development for residential use, there may have been intrusions of terrestrial carbon into the lakes. These patterns could be elucidated through paleolimnological investigation. In addition, sediment distribution is associated with wave action and boating can induce wave-like water movements, even on small lakes. The urban lakes in this study, however, restricted the use of motors, allowing only those that propel boats at speeds that do not generate waves. Nevertheless, although we have observed a strong correlation between sediment organic matter and coarse wood density, we cannot definitively rule out human-induced changes to littoral sediments in contributing to these patterns.

Implications

Most lakes in the world are net heterotrophic (Cole and others 1994, 2000; Hanson and others 2003). The respiration of carbon in lakes is concentrated in littoral habitats, which are conducive to decomposition and other biodegradation processes because they are warm and oxygenated (Pieczynska 1990b). If the presence of littoral woody debris enhances degradation processes by retaining organic carbon in the littoral zone, coarse wood may contribute, albeit to an unknown extent, to lake heterotrophy and shift the balance of carbon storage in lake sediments relative to evasion to the atmosphere.

Loss of organic matter from littoral sediments has important implications for food web structure and ecosystem function in urban lakes. Allochthonous particulate carbon inputs are generally considered to support only a minor fraction of lake productivity (for example, Saunders and others 1980; and references), yet there is some evidence that it can be vital to secondary production in some lakes (Pace and others 2004; Cole and others 2006). Terrestrially derived detritus accumulates in the shallow waters of lakes in greater amounts than living material, and in this way acts as an energy reservoir (Saunders and others 1980), fuelling microbial processes and secondary production of aquatic invertebrates that in turn support upper trophic levels. Because most lentic biota are associated to some degree with littoral habitats, it is likely that the loss of coarse wood and the resulting reduction in organic matter destabilizes lake food webs via reduced littoral secondary productivity and subsequent trophic decoupling. In particular, our results suggest a decline in the density of benthic macroinvertebrates in urban lakes, which is likely to impair the ability of urban lakes to support fish that rely heavily on benthic resources.

ACKNOWLEDGMENTS

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Attachment 3

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DEGRADATION OF LITTORAL HABITATS BY RESIDENTIAL DEVELOPMENT: WOODY DEBRIS IN LAKES OF THE PACIFIC NORTHWEST AND MIDWEST, UNITED STATES

INVASIVE SPECIES, GLOBAL POTENTIAL • SEA-LEVEL RISE, ECONOMIC IMPACT, SINGAPORE • MIRABEL FRAMEWORK, CALCAREOUS GRASSLANDS, EUROPE • EU WATER FRAMEWORK DIRECTIVE, BALTIC SEA

Report

Tessa B. Francis and Daniel E. Schindler

Degradation of Littoral Habitats by Residential Development: Woody Debris in Lakes of the Pacific Northwest and Midwest, United States

One of the least understood aspects of aquatic ecology is the role of riparian zones of lakes, and how these habitats and their functions are impacted by human development of lakeshores. We investigated the effects of residential lakeshore development on littoral coarse woody debris (CWD) distribution and on riparian forest characteristics by comparing 18 lakes in the U.S. Pacific Northwest with 16 previously surveyed lakes in the U.S. Upper Midwest. Residential development had a strong negative effect on CWD and riparian forest characteristics at both local and whole-lake scales. There was a strong positive correlation between riparian forest density and littoral CWD abundance in both regions. We found regional variation in CWD and riparian forest characteristics, mostly owing to differences in native forests. Our results suggest the role of local processes in determining CWD distribution and point to potential regional differences in littoral habitat structure associated with forest composition and lakeshore development that may have consequences for littoral-pelagic coupling in lakes.

INTRODUCTION

Increasing attention is being drawn in ecology to the linkages between habitats and ecosystems that have traditionally been considered distinct (1, 2). Associations between adjacent habitats are typically defined by inputs or exchanges of material or energy across boundaries which enhance productivity in recipient ecosystems (1). Much of this work has focused on stream, river, and marine coastal systems (3–7), in part because the riparian zone, the transition zone between terrestrial and aquatic habitats, is an especially active area in these systems for exchange of energy and material that is increasingly recognized as vital to recipient ecosystems (8, 9).

More recently, attention has been drawn to habitat coupling in lake ecosystems (10, 11). A well-understood relationship between lakes and their associated watersheds is the input of nutrients that affect in-lake primary productivity (12). In addition, inputs of exogenous particulate (13–15) and dissolved (16, 17) organic material contribute substantially to lake production. Terrestrial insects are deposited in large numbers on the surface waters of lakes, where they can become key components of fish diets (18–21). Another critical allochthonous input to aquatic ecosystems is coarse woody debris (CWD), large pieces of fallen and dead wood that represent essentially permanent features of littoral habitats of lakes in forested regions (22, 23).

The crucial role of CWD in river and stream ecosystems is well recognized (24–28) and is now a major focus of river management and restoration. In riverine systems, CWD provides physical structure that increases channel stability, alters hydrological processes, and controls the spatial distribution and extent of sediment accumulation (28, 29). In addition, CWD plays a key role in creating habitat for fishes and invertebrates by directly providing refuge as well as by forming pools that are a critical habitat for many fish species (24, 26).

Considerably less is understood about the importance of littoral CWD in lakes, though CWD accumulation can be substantial (30) and dead wood is very stable in littoral zones, remaining in standing waters for as long as 800 years (22). In lakes of the Upper Midwest, CWD density was positively correlated with growth rates of dominant fishes (31), and fish preferentially aggregate in littoral zones with CWD as compared with more simplified habitats (32, 33). In addition, CWD can represent substantial surface area on lake bottoms (28) and is a common site of macroinvertebrate colonization (34–36). Although many of its functions in littoral habitats are not fully understood, CWD is likely to be important in lakes because of its occurrence in high densities, its longevity in standing water, and what is known about the key role it plays in river and stream systems.

Human residential development of lakeshores is associated with changes in key ecosystem characteristics, including nutrient levels (37), aquatic vegetation (38), and the spatial distribution (39) and growth rates (31) of fishes. Concomitant with increasing development pressure, for at least a century, humans have systematically removed CWD from aquatic ecosystems, both directly and indirectly. Christensen et al. (30) showed that humans directly simplify the littoral zone by removing CWD and aquatic vegetation in lakes of the Upper Midwest region of the United States. In Europe, where urban areas were developed earlier than in North America, this removal has likely been occurring for much longer. However, as of yet, there has been little systematic comparison of littoral CWD characteristics, and the effects of human lakeshore development on littoral CWD distribution, among geographic regions with different forest characteristics.

Very few studies exist on the distribution and function of CWD in the littoral zones of lakes. Our purpose is to identify general patterns of riparian forest characteristics and corresponding littoral CWD distribution associated with residential development on lakeshores. To do this, we compare results from a survey of lakes in the Upper Midwest (30) with lakes in the Pacific Northwest, two regions in the United States located within distinct biomes and associated forest characteristics. (Figure 1). We expected to find regional variation in littoral habitat characteristics, specifically CWD size and spatial

Figure 1. Map of United States showing Pacific Northwest and Upper Midwest regions.

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Table 1. Vegetation characteristics of the riparian zones surrounding study lakes of the Pacific Northwest and Midwest, United States.

Region	Climax dominant	Early- or mid-climax dominants	Other major riparian tree species
Pacific Northwest	Western hemlock *Tsuga heterophylla*	Douglas-fir *Pseudotsuga menziesii* Western red cedar *Thuja plicata*	Big leaf maple *Acer macrophylla* Black cottonwood *Populus trichocarpa* Red alder *Alnus rubra*
Midwest	Eastern hemlock *Tsuga canadensis* Sugar maple *Acer saccharum*	Balsam fir *Abies balsama* Paper birch *Betula papyrifera* Red maple *Acer rubrum*	Northern white-cedar *Thuja occidentalis* Black spruce *Picea mariana* Tamarack *Larix laricina*

Sources: Pacific Northwest, (44); Midwest, (30).

distribution, associated with differences in forest composition. Not only are we interested in the effects of development at a whole-lake scale, but also in riparian-littoral dynamics occurring at more local scales, such as littoral CWD distribution on forested sites of partially developed lakes as compared with deforested sites (i.e., residential sites) on those same lakes, or sites on undeveloped lakes. Finally, we discuss some management considerations for riparian and littoral habitats, particularly in light of regionally unique patterns.

MATERIALS AND METHODS

Our study followed the methods of Christensen et al. (30), who surveyed CWD characteristics in lakes and forest composition in adjacent riparian habitats in the Upper Midwest, a region in the north-central and northeastern part of the United States, and specifically the northern areas of the states of Wisconsin and Michigan. For the current study, we sampled 18 lakes in the Pacific Northwest, a region of the northwestern United States, and specifically areas of western Washington State and southern British Columbia, Canada. The Pacific Northwest lakes are all located in the western hemlock zone of the Cascade Range and Puget Trough regions, and the forests surrounding the study lakes in the Upper Midwest (hereafter "Midwest") are mixed northern hardwood-conifer forests (Table 1). Most of the forests in both regions were cleared by European settlers by the end of the late 19th century and are currently second or third growth (30, 40).

Most of the lakes in the current study are located in the suburban cities surrounding the Seattle metropolitan area. In addition, three lakes were selected as reference systems and are located in the University of British Columbia's Malcolm Knapp Research Forest, British Columbia, Canada. These lakes lack residential development, although their associated riparian forests were also cleared at the end of the 19th century, and therefore are not without human influence. Since the initial deforestation, however, the riparian forests have remained intact and have matured, and the three lakes are therefore categorized as undeveloped.

On each lake, we established 300-m^2 rectangular sample plots measuring 30 m along the shoreline and 10 m into the riparian forest. We also sampled the 30-m stretch of lakeshore adjacent to each riparian plot. This plot size was selected because the standard residential plot on these lakes is approximately 30 m wide, and because approximately 80% of CWD originates from the first 10 m of riparian forest (41). Between four and eight sample plots per lake were selected at random, according to a stratified process such that on lakes that were partially developed, we sampled both forested and deforested sites in equal number, aiming for eight plots total (four of each type). On entirely undeveloped lakes where only forested sites existed and on lakes with fully developed lakeshores where only deforested sites existed, we sampled four plots distributed randomly along the lakeshore.

In each 300-m^2 sample plot, we counted and measured the diameter at breast height of all trees larger than 5 cm in diameter, as in Christensen et al. (30). We also counted and measured the diameter of all pieces of CWD greater than 10 cm in diameter and longer than 1 m (42) intersecting the 0.5-m depth contour in the littoral zone adjoining the sample plot. We calculated basal area for each plot as the sum of all individual tree or log basal areas, defined as basal area $= \pi \times r^2$, where $r =$ radius at breast height (trees), or where the log intersected the 0.5-m depth contour.

We compared CWD and forest characteristics at the whole-lake scale and at the local site scale. Whole-lake means were weighted by the relative amount of shoreline that was forested versus developed. We tested the effect of residential development on mean lake CWD density and basal area by using analysis of covariance with region (Pacific Northwest and Midwest) and development intensity (undeveloped, low, high) as main factors, and riparian forest density as a covariate. We also used linear regression to analyze the relationship between CWD density and basal area and residential development, using whole-lake means. One lake (Langlois Lake) was a consistent outlier in CWD density and basal area, with extraordinarily higher values for both metrics, and was therefore removed from comparisons at the whole-lake scale. At the local site scale, we used analysis of variance with Sheffé's post hoc test ($\alpha = 0.05$) by using site type (undeveloped, developed-forested, and developed-deforested) as main factors and CWD density, CWD size, riparian tree density, and riparian tree size as response variables. To facilitate regional comparisons, we used published data from Christensen et al. (30) for whole-lake mean values and raw data from this earlier survey for tests of site type effects.

RESULTS

Study lakes were selected to span a gradient of residential development intensity (Table 2). Lakes in the Pacific Northwest (PNW) were binned to match the development intensity categories of Christensen et al. (30) according to: undeveloped $= 0$ houses km^{-1} of shoreline ($n = 3$); low $= 1$–10 houses km^{-1} ($n = 3$); high $=$ more than 10 houses km^{-1} ($n = 12$; Table 2).

We found a significant negative effect of residential development on CWD density and basal area in the PNW, and patterns of CWD distribution were tightly coupled to riparian Forest characteristics at the whole-lake scale, similar to what has been observed in lakes of the Midwest (MW). We found significant covariation between CWD density and riparian tree density in the PNW, as was found previously in the MW.

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Table 2. Characteristics of Pacific Northwest lakes sampled for coarse woody debris and riparian forest distribution.

Lake	Location	Surface area (ha.)	Residence density (houses km^{-1})	Development intensity
Gwendoline	British Columbia	13	0	Undeveloped
Loon	British Columbia	48.6	0	Undeveloped
Marion	British Columbia	13.3	0	Undeveloped
Langlois	Washington	15.8	1.1	Low
Fenwick	Washington	9.7	3.7	Low
Armstrong	Washington	12.2	6.2	Low
Neilson (Holm)	Washington	7.7	15.1	High
Silver	Washington	72.9	15.5	High
Wilderness	Washington	27.9	16.6	High
Boren	Washington	7.3	19.1	High
Geneva	Washington	10.5	22.8	High
Beaver #2	Washington	25.1	26.4	High
Shadow	Washington	19.8	27.3	High
Meridian	Washington	60.7	30.8	High
Devils (Lost)	Washington	5.3	36	High
Shady	Washington	8.5	40	High
Star	Washington	14.2	40.6	High
Angle	Washington	40.5	46.6	High

Note: All lakes are located between 47–49N latitude and 121 - 122W longitude. Characteristics of Upper Midwest lakes previously sampled are given in Christensen et al. (30).

Maximum riparian forest density was larger in the PNW than in the MW (Fig. 2). CWD basal area also covaried significantly by riparian tree basal area in the PNW, as in the MW (Table 3 and 4; Fig. 3). CWD density varied significantly by development at the whole-lake scale in both regions (Table 3 and Table 4; Fig. 4). Mean lake CWD densities tended to be lower on undeveloped lakes in the PNW than undeveloped lakes in the MW (Table 4; Fig. 4A). We also found a significant effect of development on CWD basal area in both regions, and CWD basal area covaried significantly with riparian forest basal area (Table 3 and Table 4; Fig. 4B). Maximum CWD basal area was greater in the Pacific Northwest (46.2 m^2 km^{-1} in the Pacific Northwest versus 19.4 m^2 km^{-1} in the Midwest), as was maximum riparian forest density (Table 4; Fig. 4B).

We found negative correlations of both CWD density ($p < 0.001$, $r^2 = 0.60$) and basal area ($p < 0.001$, $r^2 = 0.57$) with residential density in the Pacific Northwest, as has previously been shown in the Midwest (Fig. 4). Riparian forest density was also negatively correlated with lakeshore residence density in both regions (Pacific Northwest: $p < 0.001$, $r^2 = 0.72$; Midwest: $p = 0.002$, $r^2 = 0.50$; data not shown). In general, there was more CWD in Pacific Northwest lakes than in the Midwest (Table 4; Fig. 4A), and CWD persisted at higher development intensities in the Pacific Northwest than in the Midwest (Fig. 4A).

At the local site scale, we found a significant effect of

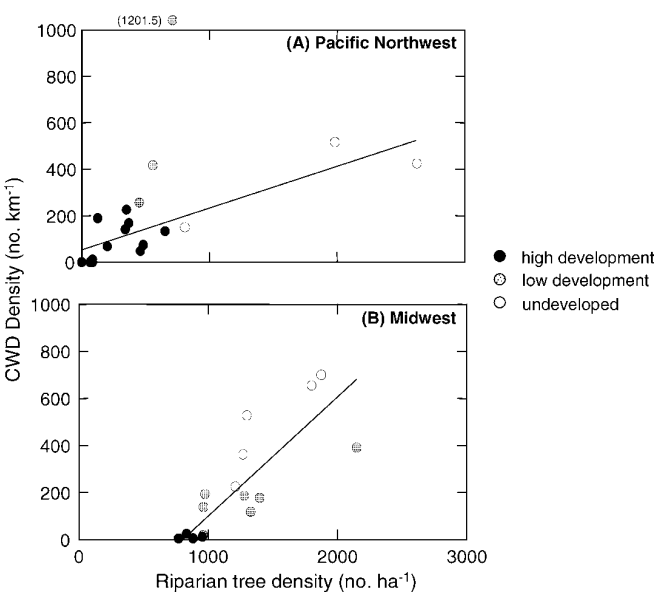

Figure 2. Coarse woody debris (CWD) density as a function of riparian tree density for Pacific Northwest (A) and Midwest (B) lakes. Circles represent whole-lake means. Open circles indicate undeveloped lakes; hatched circles, lakes with low resident density (0.1–10 residences km^{-1}); and solid circles, lakes with high resident density (>10.0 residences km^{-1}). Midwest data are redrawn from Christensen et al. (30).

Table 3. Summary of analysis of covariance results showing effect of human residential development level (high, low, undeveloped) and region (Pacific Northwest, Midwest) on whole-lake coarse woody debris (CWD) density and basal area with riparian tree (RT) characteristics as covariates.

Response	Main factor	Covariate	*F* ratio	*p* value
CWD density (pieces km^{-1} of shoreline)	Development		4.2	0.03
	Region		9.6	0.005
	Region × development		15.2	<0.001
		RT density	27.2	<0.001
CWD basal area (m^2 km^{-1} of shoreline)	Development		7.8	0.002
	Region		12.0	0.002
	Region × development		0.6	0.5
		RT basal area	7.4	0.01

Table 4. Least squares means from analysis of covariance showing effects of lake development and region on whole-lake coarse woody debris (CWD) density and basal area, with riparian tree characteristics as covariates, in lakes of the Pacific Northwest and Midwest, United States.

Region	Density (pieces km^{-1} of shoreline)	Basal area (m^2 km^{-1} of shoreline)
Undeveloped	252.4	18.3
Low development	255.1	11.8
High development	128.3	4.6
Pacific Northwest	272.0	16.7
Midwest	151.9	6.4
Pacific Northwest/undeveloped	150.3	24.5
Midwest/undeveloped	354.4	12.0
Pacific Northwest/low development	429.0	18.0
Midwest/low development	81.2	5.6
Pacific Northwest/high development	236.5	7.7
Midwest/high development	20.0	1.6

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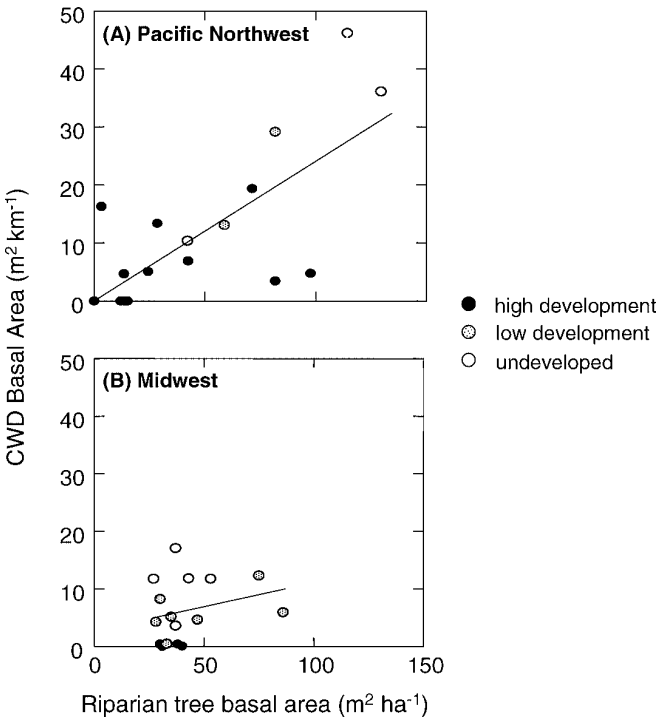

Figure 3. Coarse woody debris (CWD) basal area as a function of riparian tree basal area for Pacific Northwest (A) and Midwest (B) lakes. Symbols as in Fig. 1. Midwest data are redrawn from Christensen et al. (30).

development (undeveloped, developed-forested, and developed-deforested sites) and region on CWD density (Fig. 5; Tables 5 and 6). Post hoc tests showed that in the Pacific Northwest, CWD densities were highest on undeveloped and developed-forested sites, and not significantly different from each other. In contrast, post hoc tests showed that in the Midwest, CWD density on undeveloped sites was higher than on developed-forested sites, and lowest on developed-deforested sites. Riparian tree density also varied by development type in both regions (Tables 5 and 6). Post hoc tests showed that in the Pacific Northwest, riparian forest density was highest on undeveloped sites, whereas in the Midwest, undeveloped sites and developed-forested sites had similar forest densities.

CWD size (basal area) did not vary significantly with local site development in the Pacific Northwest, similar to the pattern previously found in the Midwest (Tables 5 and 6; Fig. 6). Post hoc tests showed that mean CWD size was greater in the Pacific Northwest than in the Midwest. Riparian tree size varied significantly among site types, and between regions (Tables 5 and 6; Fig. 6). Mean riparian tree size was greater on sites with dwellings (developed-unforested) than on either undeveloped or developed-forested sites in both regions (Fig. 6). Riparian trees were significantly larger in the Pacific Northwest than in the Midwest on sites of all development type (Fig. 6; Tables 5 and 6).

DISCUSSION

Consistent with other published findings for the United States Upper Midwest (30) and Wisconsin State lakes (43), we found that in the Pacific Northwest, residential development around lakeshores is associated with a striking loss of CWD in littoral zones and reduction of riparian forest. As expected, we found that replacement of native forests with homes, lawns, and other residential structures corresponds to lakewide alterations of riparian and littoral habitats. We also found that development impacts littoral and riparian habitats at finer scales, associated with settlement intensity on individual sites. These effects varied between the Pacific Northwest, the focus of the current study, and previously published results for lakes in the Midwest, in ways largely indicative of differences in native forests, human influences on lakeshore habitats, and coupling between riparian and littoral habitats.

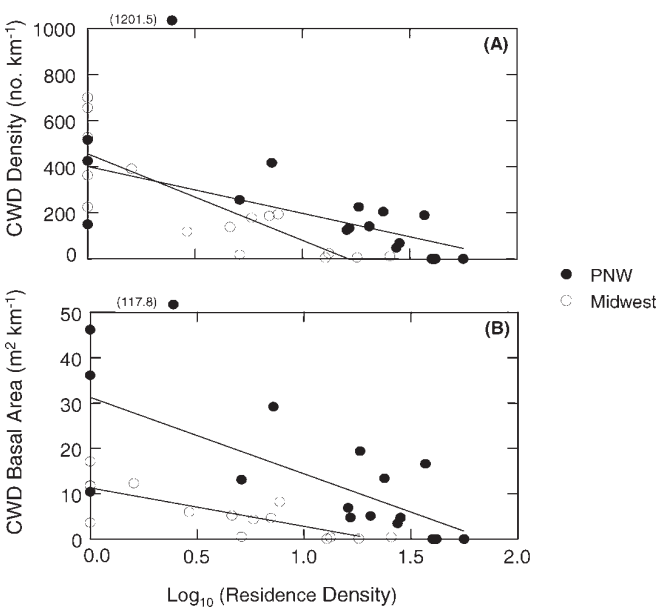

Figure 4. Coarse woody debris (CWD) density (A) and basal area (B) as a function of shoreline residential density (logarithm of number of residences km^{-1} of shoreline). Circles represent whole-lake means. Solid circles indicate Pacific Northwest lakes; open circles, Midwest lakes. Midwest data are redrawn from Christensen et al. (30).

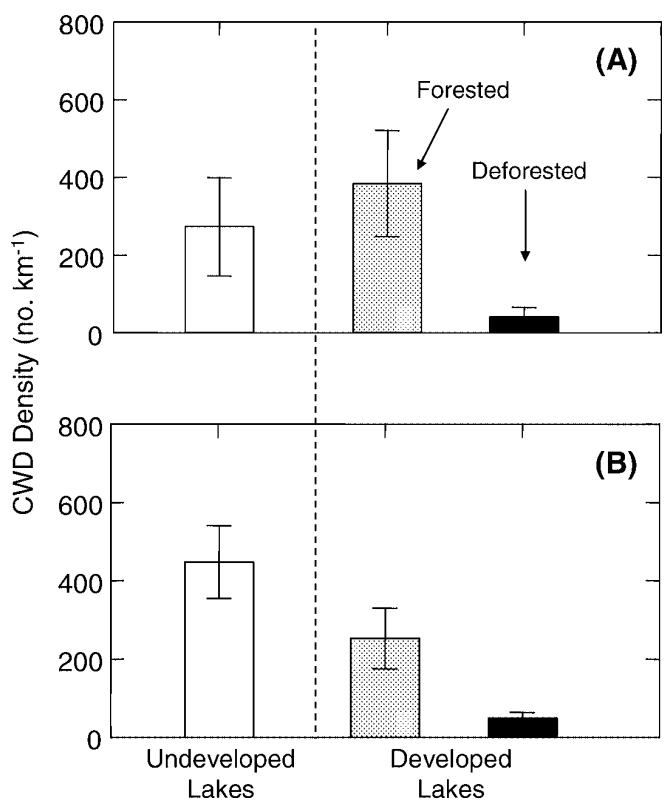

Figure 5. Coarse woody debris (CWD) density in the Pacific Northwest (A) and Midwest (B) for all lakes. Open bars indicate plots on undeveloped lakes; hatched bars, forested plots on developed lakes; and solid bars, residential (deforested) plots on developed lakes. Error bars represent 95% confidence intervals. Midwest data are from Christensen et al. (30).

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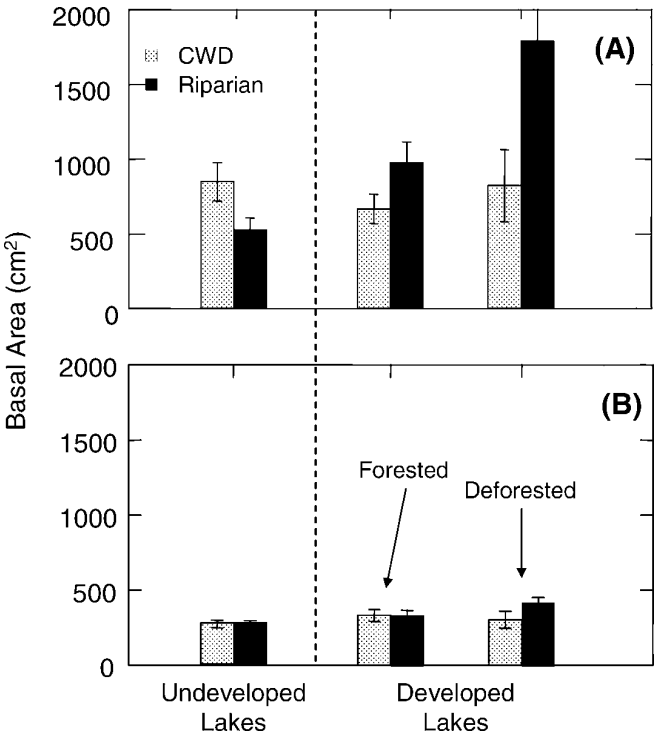

Figure 6. Coarse woody debris (CWD) and riparian tree size in the Pacific Northwest (A) and Midwest (B) for all lakes. Hatched bars indicate CWD; solid bars, riparian trees. Shown are means across all undeveloped lakes, forested plots on developed lakes, and residential (deforested) plots on developed lakes. Error bars represent two standard errors. Midwest data are from Christensen et al. (30).

Our results highlight some of the differences in native riparian forests between the Pacific Northwest and the Midwest. We found that riparian forests of the Pacific Northwest have fewer, larger trees, whereas riparian habitats in the Midwest have more, smaller trees. The larger tree size on Pacific Northwest lakeshores is likely due to the dominance in the Pacific Northwest of larger climax tree species, such as Douglas fir and western red cedar (44). Even when excluding Pacific Northwest lakes with higher development intensity than observed in Christensen et al. (30), the same patterns are observed (data not shown), allowing us to dismiss the hypothesis that this is a development-intensity, and not a regional, effect.

In lakes of both the Midwest and the Pacific Northwest, we found strong coupling between riparian forest composition and littoral habitat structure. The regional differences in forest composition we observed translate into regional variation in CWD size and volume, such that forests in the Pacific Northwest generate larger and greater overall volume of littoral CWD. Regionally distinct forest composition may therefore have consequences for aquatic biotic associations limited by availability of CWD surface area, such as aggregations of macroinvertebrates on CWD in response to surface biofilm (34, 45, 46).

We also found significant interactions between region and development in the relationship between CWD density and riparian forest density. In the Pacific Northwest, CWD density was lower than expected on undeveloped lakes, which had the highest riparian stem density of all lakes sampled. In contrast, in the Midwest, CWD density patterns directly reflected riparian forest density patterns, suggesting a strong relationship between riparian and littoral habitats on undeveloped lakes. One potential explanation for this regional variation is that in the Pacific Northwest, riparian stem density of newer growth forests on undeveloped lakes may not yet be reflected in littoral habitats, where the legacy of large old-growth fallen trees is retained in relatively fewer, larger logs. Likewise, smaller riparian trees on Midwest lakes may be more sensitive to mechanisms that generate CWD, such as windthrow and beaver foraging. This is supported by the significant relationship between CWD basal area (i.e., total littoral CWD volume) and riparian forest basal area in both regions. Alternatively, CWD may be lost from littoral habitats in the Pacific Northwest undeveloped lakes more rapidly than in the Midwest. One such mechanism could be size of lake, because larger lakes with greater fetch may generate wave action and displace CWD, although mean surface area for undeveloped lakes in each region was virtually identical (Pacific Northwest 25.0 ha; Midwest 26.0 ha).

Table 5. Summary of analysis of variance results showing effects on a local site scale of development (undeveloped, developed-forested, developed-deforested) and region (Pacific Northwest, Midwest) on coarse woody debris (CWD) and riparian tree (RT) density and basal area.

Dependent variable	Effect	*F* ratio	*p* value
CWD density (pieces km^{-1} of shoreline)	Region	12.8	<0.001
	Development	61.5	<0.001
	Region × development	8.0	<0.001
CWD basal area (cm^2 piece^{-1})	Region	0.26	<0.001
	Development	60.8	0.78
	Region × development	1.3	0.27
RT density (trees ha^{-1})	Region	13.6	<0.001
	Development	60.3	<0.001
	Region × development	11.9	<0.001
RT basal area (cm^2 tree^{-1})	Region	42.2	<0.001
	Development	14.9	<0.001
	Region × development	8.9	<0.001

Table 6. Least squares means from analysis of variance showing local effects of development and region on coarse woody debris (CWD) and riparian tree (RT) density and size in lakes of the Pacific Northwest (PNW) and Midwest (MW), United States.

Region	CWD density (pieces km^{-1} of shoreline)	CWD basal area (cm^2 piece^{-1})	RT density (pieces ha^{-1})	RT basal area (cm^2 tree^{-1})
Undeveloped	252.4	490.3	1619.4	420.3
Developed-forested	255.1	444.4	1140.9	751.3
Developed-deforested	128.3	484.0	492.3	1230.8
PNW	272.0	715.1	925.0	1214.3
Midwest	151.9	230.7	1243.4	387.4
PNW/undeveloped	150.3	775.8	1805.6	775.8
MW/undeveloped	354.4	204.8	1433.3	554.5
PNW/developed-forested	429.0	622.0	808.8	1110.2
MW/developed-forested	81.2	266.8	1472.9	392.4
PNW/developed-deforested	236.5	747.5	160.7	1978.2
MW/developed-deforested	20.0	220.5	823.8	483.5

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In addition to the striking effects of development on littoral and riparian characteristics at the whole-lake scale, we also found localized variation in littoral CWD and riparian forest distribution and size associated with the presence of residential structures. On Midwest lakes, CWD densities on developed-forested sites were intermediate to those on undeveloped and residential sites. In contrast, in the Pacific Northwest CWD densities on developed-forested sites were equivalent to those on undeveloped sites. It is unclear why this variation between regions exists, although it may be associated with higher overall CWD densities in undeveloped Midwest lakes, some hypotheses for which were presented above. In both regions, however, CWD densities on developed-forested sites were higher than on developed-unforested sites. Given the usual persistence of CWD in aquatic ecosystems (22), the highest rates of CWD loss on developed sites most likely occur via direct removal by people. CWD densities on forested sites of developed lakes suggest that CWD continues to be delivered to littoral zones from locally forested habitats, and that individual residents have a local effect on littoral CWD, either directly, through its removal or indirectly, through deforestation. The manifestation of reduced CWD source material may occur on a longer timescale than the direct removal of CWD from littoral zones, but nevertheless, deforesting lakeshores likely results in a reduction of CWD inputs to shallow waters (41) and severely diminishes the future capacity for restoration of these ecosystems.

We found that human development also altered the size distribution of littoral CWD and riparian trees in both regions at the local scale. Riparian trees were largest on residential plots of developed lakes in both regions, although the difference was most dramatic in the Pacific Northwest. This "trophy tree" phenomenon presumably occurs when developers or residents preferentially leave larger trees on lakeshore property for aesthetic purposes. Often, these trophy trees are nonnative, ornamental species with different successional and litterfall patterns than native trees (T. Francis pers. observation). Sociological surveys in these two regions could clarify the mechanisms that produce this difference in land-use patterns between the Midwest and the Pacific Northwest.

Our results suggest several potential disruptions to the coupling between littoral CWD and riparian trees by humans, including deforestation, high grading for larger trees, and directly removing CWD, all of which can potentially alter ecological function in littoral habitats (28, 47). Although all the functions of CWD in littoral habitats are not known, it is clear that CWD provides key habitat for macroinvertebrates and fishes (32, 34, 36, 46). Alterations to littoral structure such as CWD distribution likely affect benthic communities and trophic interactions. Changes in benthic communities may have ecosystem-wide consequences owing to the tight coupling between benthic and pelagic habitats (10). Thus, not only should we expect consequences for littoral and benthic communities associated with residential development, but also natural and human-induced variation in forest and CWD composition raises questions about associated differences in the relationships between riparian and littoral habitats and benthic-pelagic coupling.

The regional differences outlined here have implications for management strategies related to the maintenance of ecosystem functions in lakes. Actions to retain CWD in littoral zones could be designed with targets for either the number of CWD pieces per length of shoreline or the overall volume of wood in littoral zones. Comparison of the data from these two regions demonstrates that this choice could have different implications in each region. Undeveloped lakes in the Midwest had higher densities of CWD than undeveloped lakes of the Pacific Northwest, but lower overall volumes of CWD. A single strategy based on either the density or total volume would not be appropriate for both regions; rather, management schemes should address both metrics of littoral CWD distribution. In addition, management strategies associated with residential development densities should be tailored to the unique associations between residence density and CWD in each region. In the Pacific Northwest, total loss of CWD occurred at residential densities greater than 20 houses km^{-1}. However, the lowest levels of CWD in the Midwest occurred at much lower residential densities (12 houses km^{-1}). Thus, lake management strategies incorporating development targets should also be regionally defined.

Finally, the decoupling of terrestrial-aquatic linkages by residential development of lakeshores has ecosystem consequences that will likely persist for centuries. Most CWD is produced by periodic disturbance (24). Humans have interrupted the inputs of littoral CWD not only by removing riparian sources of CWD, as seen here, but also by disrupting this disturbance cycle (48). On the basis of estimates of CWD input rates to rivers and streams, where such estimates actually exist, returning littoral CWD levels to those observed on undeveloped lakes will take many decades. Furthermore, these estimates are conservative because they assume the presence of intact riparian forest and were calculated for fluvial systems where CWD inputs are greater as a result of the more transient nature of riparian vegetation and steeper riparian slopes. Active restoration practices, such as directly adding CWD to lakes, may accelerate this recovery but such strategies would be financially costly. Rather, given the rapid pace of human development, the most prudent management actions should now focus on active protection of intact littoral habitat where it is still present (49).

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October 14, 2009 Planning Commission Meeting Agenda Memo - Attachment 4
Shoreline Master Program Update Project Planning Commission Schedule

DATE	TOPIC
March 12, 2008	• Brief introduction to SMP- highlighting major steps
July 23, 2008	• Current status, next steps and key future milestones of Shoreline Master Program (SMP) Update
Sept. 10, 2008	• Ian Stewart- EMC Research- results of opinion survey • Current project status, survey's key findings, and details of upcoming shoreline boat tour September 20th
Jan. 28, 2009	• Ian Stewart EMC Research-summary of findings from City's formal focus group with shoreline property owners conducted in November 2008. • Framework for development of shoreline policies • Presentation by Dan Nickel of The Watershed Company- purpose, methods, and conclusions of Shoreline Inventory and Analysis Report • Key components of proposed work plan- Commission's concurrence with proposed approach • Current project status, character of focus group, and details of Shoreline standards included in Critical Area Code
Feb. 25, 2009	• Potential shoreline environment designations • Direction from Commission on range of designations appropriate for Bellevue and areas identified in the Inventory and Analysis warranted special treatment
May 27, 2009	• Continued discussion of potential shoreline environment designations • Information regarding proposed "alternative" designations for sites
July 8, 2009	• Legal department presentation - Shoreline Management Act with focus on Anacortes case • Details of Critical Areas code • Site specific environment designations-revised vision for Bellefield Office Park Option #2 and summary of past discussions for alternative designations • Affected property owners input summary
July 22, 2009	• Policy review necessary for the required SMP update • Central role of Guidelines • Key components of a master program • Way to move forward updating the SMP • Policy work including purpose language and management policies for each adopted environment designation
September 23, 2009	• Strategy for moving forward • Introductory science briefing- Dan Nickels The Watershed Company • Tracking and providing information to the PC
October 14, 2009	• City of Bellevue Storm and Surface Water Management - Denny Vidmar Director Utilities Department
October 28, 2009	• **Affect of Lakeshore Development on Aquatic Ecosystems- Tessa B. Francis PhD NOAA Fisheries**
November 4, 2009	• The Current Status of Kokanee in the Greater Lake Washington Watershed- David St. John, King County Department of Natural Resources and Parks

October 14, 2009 Planning Commission Meeting Agenda Memo - Attachment 4
Shoreline Master Program Update Project Planning Commission Schedule

DATE	TOPIC
November 18, 2009	• Behavior and Habitat use of Chinook Salmon - Roger Tabor and/or Mark Celedonia • Shoreline stabilization and the importance of shoreline vegetation.- Jose Carrasquero, Herrera Environmental Consultants Inc.
December 9, 2009	• Tentative Panel/Presentation from related regional and state organizations engaged in similar SMA goal efforts (examples- Army Corps of Engineers, NOAA Fisheries, WDFW, WRIA 8, Puget Sound partnership)
December 2009/January 2010	• Regulatory Outcomes- TBD-Planning Commission Direction
Winter/Spring 2010	• Additional Regulatory and Policy Discussion
Spring /Summer 2010	• Cumulative Impacts Analysis and Restoration Plan
Summer 2010	• Local Approval
Summer/Fall 2010	• State Approval