CHAPTER 9 THE EFFECTS OF WATERSHED DEVELOPMENT ON WATER QUALITY AND SOILS

by Richard R. Horner, Sarah Cooke, Lorin E. Reinelt, Kennneth A. Ludwa, Nancy Chin and Marion Valentine

INTRODUCTION

This chapter emphasizes water and soil quality in wetlands with significant urbanization in their watersheds. Like other chapters in this section, its purpose is to characterize particular elements of Puget Sound Basin freshwater wetlands having urbanized watersheds. The urbanized cases were divided into two major categories. The "treatment" group included the wetlands whose watersheds had a more than 10% rise in urbanization between 1989 and 1995. Conversely, the "control" category consisted of wetlands whose watersheds experienced a less than 10% increase in urban land cover between 1989 and 1995. The urban control wetlands were subdivided into two further classifications: (1) the most highly urbanized sites (H) had watersheds that were both ≥ 20% impervious and ≤ 7% forest by area; and (2) moderately urbanized wetlands (M) had watersheds that were 4-20% impervious and 7-40% forested by area. The nonurbanized category (N), with both < 4% impervious land cover and ≥ 40% forest, made up the balance. This latter category is emphasized in Chapter 2 but will be mentioned in this chapter at times for comparison. Table 1 of Section 1 gives characteristics of the individual watersheds.

This chapter first describes water quality conditions in urban control wetlands, and then discusses changes in these conditions in treatment wetlands. It then proceeds to cover soil characteristics in a similar way. Chapter 2 covers the methods with which the data were collected. Tables 1, 2, and 4 of Chapter 2 summarize water quality results for the urban control wetlands as well as the nonurbanized cases. Tables 5 and 6 in that chapter perform the same function for the soils data. These tables are not repeated in this chapter but are referenced here several times.

As has been stated elsewhere in this volume, the research program concentrated on palustrine wetlands of the general type most prevalent in the lower elevations of the central Puget Sound Basin. The results and conclusions presented here are probably applicable to similar wetlands to somewhat to the north and south of the study area but may not be representative of higher, drier, or more specialized systems, like true bogs and “poor” (low nutrition) fens.

THE EFFECTS OF WATERSHED DEVELOPMENT ON WATER QUALITY

This section first profiles the urban control wetlands, both moderately and highly urbanized, using the statistical summary data in Chapter 2, Tables 2-1, 2-2, and 2-4. Following the profiles is a more general summary of other applicable findings from the research.

Moderately Urbanized Wetlands

A water quality portrait of Puget Sound Basin lowland palustrine wetlands moderately affected by humans would show slightly acidic (median pH = 6.7) systems with DO often well below saturation, and in fact sometimes quite low (< 4 mg/L). Dissolved substances are fairly high relative to nonurbanized wetlands (median conductivity about three times as high) but somewhat variable. Suspended solids are only marginally higher than N wetlands but, like them, quite variable. Median total dissolved nitrogen concentrations (the sum of ammonia, nitrate, and nitrite) are more than 20 times as high as dissolved phosphorus, a ratio very similar to the nonurbanized wetlands but with higher magnitudes in both cases. Again, plant and algal growth is generally limited by P, rather than N. TP at the median level is more than twice as high in the M compared to the N wetlands (70 µg/L). The median fecal coliform concentration is close to the 50 CFU/100 mL criterion applied as a geometric mean to lakes and the highest class of streams by Washington state water quality criteria. This quantity is highly variable in all wetlands, most extremely so in those of the M class. More than half of the individual FC values for moderately urbanized flow-through wetlands exceeded the maximum 200 CFU/100 ml criterion applied to the lowest class streams (however, their geometric mean may not do so). As with N wetlands, both mean and median heavy metals concentrations in the moderately urbanized sites are in the low parts per billion range, with standard deviations just about identical to the means. The median lead concentration, however, is close to the chronic water quality criterion set for lakes and streams having hardness of 50 mg/L as $CaCO_3$.

In summary, the following general statements can be made to characterize the water quality of Puget Sound Basin lowland palustrine wetlands in a moderately urbanized state:

> These wetlands are highly likely (≥ 71% of cases observed) to have median conductivity > 100 µS/cm but median TSS in the range 2-5 mg/L, NH_3-N < 50 µg/L, and total Zn < 10 µg/L.
>
> Moderately urbanized wetlands are highly likely (71% of cases) to have median fecal coliforms < 50 CFU/100 mL, but also to have many individual measurements above 200.
>
> They are highly likely (100% of cases to have TP > 20 µg/L and likely (57% of cases) to have TP > 50 µg/L and NO_3+NO_2-N < 100 µg/L. The latter variable is highly likely (86% of case) to be < 500 µg/L.
>
> The pH and DO in these wetlands are unpredictable from consideration of urbanization status alone, being dependant on other factors.

Highly Urbanized Wetlands

Highly urbanized wetlands are harder to profile because of the small set of only two in the control group. Also, both of these wetlands are the flow-through type, not giving any picture of how morphology might affect the conclusions. What can be said is the following:

> There is some tendency for these wetlands to be the closest to neutral in pH among the three urbanization status categories. They tend to fall in the same region as the other classes in dissolved oxygen.

They most likely would have median conductivity > 100 µS/cm.

Like the other two classes, highly urbanized wetlands are very likely to have median NH_3-N < 50 µg/L.

Unlike the other two classes, they are very likely to have median NO_3+NO_2-N > 100 µg/L and TP > 50 µg/L.

These sites are likely, but somewhat less than the other categories, to have autotrophic growth limited by phosphorus.

They are likely to exceed the 50 CFU/100 mL level of fecal coliforms.

These wetlands have a higher tendency than the other categories to have Zn > 10 µg/L but in most instances still not to exceed the chronic criterion of 59 µg/L for relatively soft waters.

Other Findings

In a synoptic study of 43 urban and 27 nonurban wetlands during 1987, before routine sampling began, the program found that FC and enterococcus were significantly higher in urban wetlands (Horner et al. 1988). Although the mean counts for both types of bacteria were within water quality standards, bacteria substantially exceeded standards in wetlands in high density areas that showed evidence of human intrusion. The watersheds of most of the wetlands in which bacteria also exceeded standards were in watersheds characterized by low density residential development, while some of the watersheds of the remainder of the wetlands had high density residential development. None of the watersheds with bacteria in excess of standards were dominated by commercial development. Other than pH, FC count was the only water quality variable measured by the survey.

After four years of regular data accumulated, a major effort was undertaken to relate water quality conditions to watershed and wetland morphological circumstances. In this work it was found that certain water quality parameters varied in response to the changes in watershed wetland characteristics that can accompany urbanization (Ludwa 1994). The characteristics used as independent variables in this analysis included (1) percent forest cover, (2) percent total impervious area, (3) percent effective impervious area (the area actually linked to a storm drain system), (4) the ratio of wetland to watershed area, (5) the ratio of forest to wetland area, (6) wetland morphology (open water or flow-through), and (7) outlet contstriction. These measures may be expressed as either continuous (ranges) or categorical (binary or ternary) variables. Multivariate linear regressions were used to determine if there is an adequate relationship between these characteristics and water quality parameters. If there is such a relationship, the equations could be used to analyze probable changes in water quality following development.

Specific watershed land uses and wetland morphological values were significantly associated with most water quality values (Ludwa 1994). The dependent water quality variables exhibiting the best associations and most correctly predicted when verified with a portion of the data set held aside for verification were conductivity, pH, and TSS (Ludwa 1994). Pollutants that are often adsorbed to particulates, specifically TP, Zn, and FC, showed similar degradation across key levels of the independent variables. Conductivity, TSS, FC, and enterococcus degraded the most consistently between more highly developed watersheds and those that were moderately urbanized or rural (Taylor et al. 1995). Conductivity, TSS, Zn, DO, and FC varied by the greatest amounts (Ludwa 1994).

Based on program data from 1988-93, percent forest cover was the best predictor of water quality for Pacific Northwest palustrine wetlands, followed by percent total impervious area, forest-to-wetland areal ratio, and morphology (Ludwa 1994). All variables except NO_3+NO_2-N were higher in wetlands with no forest in their watersheds compared to those in watersheds with at least 14.7% forest cover (Taylor et al. 1995). Conductivity, TP, and FC rose significantly when the percentage of impervious surface exceeded the values of 3.5% and 20% (Taylor et al. 1995). Forest-to-wetland areal ratio strongly influenced conductivity, TSS, NO_3+NO_2-N, TP, SRP, and FC, where the ratio was less than 7.2 (Taylor et al. 1995). Conductivity, TSS, NO_3+NO_2-N, TP, SRP, and FC, had significantly higher means in relatively channelized wetlands, although it should be noted that these results may have been influenced by extraneous factors (Taylor et al. 1995). Outlet constriction and wetland- to-watershed areal ratios had inconsistent roles in influencing water quality (Taylor et al. 1995). It should be noted that because the breakpoint values are expressed as fixed ranges, and it is unknown if thresholds exist or what they are, it is entirely possible that other water quality constituents also vary significantly with characteristics of urbanization on a continuous basis (Taylor et al. 1995).

The analysis indicated that, for similar watersheds in the region, there is a definite degree of deforestation and development above which average wetland water quality will become degraded. However, if amounts of forest remain above some minimum level, water quality will comply with criteria. It should be noted that because extremes in water quality often have greater impacts on biological resources than average conditions, attention should also be given to the relationships of conditions with minimum and maximum values. Minimums and maximums were found to vary widely across urbanization and morphological levels, so that entire ranges of water quality variables shift significantly to degraded conditions.

Ludwa (1994) found that the strongest regression relationships were for mean, maximum, and minimum conductivity, TSS, and DO. In view of the correlation coefficients, urbanization was consistently related to all water quality values except NH_3-N and SRP. The strong regressions of TSS and conductivity with urbanization suggested that an increase in watershed imperviousness will facilitate the movement to wetlands of runoff containing inorganic particulate and dissolved matter. Total suspended solids and conductivity are directly and indirectly harmful to wetland biological communities. Wetland morphological factors had similar effects, although they were less consistent for outlet constriction.

Predictions were generally better for mean and maximum values, since these values exhibited more variability than minimum values from site to site. The choice of factors to be included in the regression equations was manipulated to improve predictive value, although the process was somewhat subjective. Wetland-to-watershed areal ratio was the most frequently used factor. Although little can be done to affect this ratio other than by changing wetlands physically or by diverting inflows, the importance of this ratio does not suggest that development or deforestatation are unimportant. To the contrary, where a wetland covers a smaller portion of a watershed, the effect of deforestation may be magnified. Effective impervious area, which expresses how much land is actually drained by a storm drainage system, had more predictive power than total impervious area. However, there was no consistent relation between outlet constriction and water quality. Categorical predictors seemed to have slightly better predictive value than continuous ones, and are also simpler to use.

The crucial values for water quality lie between 4 and 12% for total impervious area and 0 to 15% for forested area. Theory and observations in other regional ecosystems (e. g., Horner et al. in press) have demonstrated that there is likely to be a continuous and relatively rapid decline in various measures of ecosystem "quality" as forest begins to decrease in favor of impervious surfaces. As conversion progresses the decline is likely to slow in rate but to continue. Therefore, with a continuous pattern of variation normally prevailing, numerical values should not be regarded as thresholds but as points where degradation becomes evident and demonstrable, and where standards generally accepted as necessary to support biota probably will not be met, at least at times.

Ludwa recommended that total impervious area in Pacific Northwest watersheds with strong wetland protection goals be not more than 10% and that a forest cover of at least 15% be maintained. Whether more effective implementation of urban runoff best management practices would permit these thresholds to be shifted toward more urbanization is a matter only for conjecture. However, development and deforestation will ultimately have to be limited if high quality and well functioning wetland systems are to be preserved. Channelized sites usually had lower water quality, hence a shift to more channelized conditions intentionally or by inadvertant flooding resulting from increased urbanization should be avoided.

Treatment Wetlands

The treatment wetlands studied by the program were: Big Bear Creek 24 (BBC24), East Lake Sammamish 61 (ELS61), Jenkins Creek 28 (JC28), North Fork Issaquah Creek 12 (NFIC12), and Patterson Creek 12 (PC12). Their watersheds experienced increases of urbanization in the range of 10.2 to 10.5% in three of the five cases (JC28, ELS61, and PC12), 42.2% for BBC24, and 100% in the case of NFIC12. The most common change in land use was from forest to single family residential, a development pattern typical of the early stages of urbanization (Chin 1996). Table 9-1 shows land cover in 1995 and the changes since 1989.

This distribution of changes gave an opportunity to observe the relative effects with substantial compared to more limited watershed alterations. The timing of development in relation to the program's schedule also offered the chance to observe effects during the construction-phase, when soils are often bare for long intervals, versus the subsequent period when areas finished with construction are restabilized.

Table 9-1. Land cover in 1995 and the changes from 1989.

	Forest		Impervious Surface	
Wetland	1995(%)	Change(%)	1995(%)	Change(%)
JC28	19.8	-14.6	20.6	+0.6
ELS61	3.7	+1.2	10.6	+5.5
PC12	64.7	-10.5	6.8	+1.7
BBC24	47.4	-42.1	10.6	+7.2
NFIC12	0.0	-100.0	40.0	+38.0

In terms of the urbanization groupings used to classify the control wetlands, after the development that occurred through 1995 in the treatment watersheds, NFIC12 would be categorized as H and all of the others as M. Morphologically, JC28 is a flow-through type, and the remainder are all open water.

Observations for Individual Treatment Wetlands

Wetlands in urbanizing watersheds are especially vulnerable to erosion during the construction phase of development. Total suspended solids concentrations often increase greatly during such periods, but return to approximately pre-development levels as bare land is covered by structures and vegetation. During periods of construction, mean TSS values increase more dramatically than median values because of the influence of especially high concentrations. For instance, the ELS61 wetland recorded a median TSS concentration of 10.4 mg/L in 1989 and had a maximum concentration of 59 mg/L in August, as a result of construction site runoff. An increase in TSS at JC28 in 1989 was also linked to land disturbances. At both of these sites, TSS declined in the following year.

Elevated sediment in runoff from construction sites also corresponds to increases of concentrations of other pollutants, especially phosphorus and nitrogen), that are contained in soils (Novotny and Olem 1994). Subsequent to construction, application of fertilizers can further increase nutrient concentrations in runoff. In the JC28 wetland, land disturbances, including expansion of an adjacent golf course in 1989 marked the commencement of a regime of higher nutrient concentrations. Median NO_3+NO_2-N and SRP values increased by 63% and 96%, respectively, between 1988 and 1989, and continued to climb steadily from 1990 to 1995. The initial increases probably resulted from land disturbance, with the subsequent rises attributable to fertilizer runoff from the golf course. Mean NH_3-N also also rose sharply in 1989, with a maximum value of 619 μg/L, and median NH_3-N was higher in subsequent years. More than half of the NO_3+NO_2-N readings exceeded 500 μg/L.

At the ELS61 wetland, NH_3-N and NO_3+NO_2-N initially rose in 1989, but declined in 1993, although not to predevelopment levels. Concentrations of NH_3-N climbed again in 1993, while NO_3+NO_2-N greatly increased in 1995. Many NH_3-N and NO_3+NO_2-N concentrations exceeded 100 and 500 μg/L, respectively, during these years. Average SRP and TP concentrations were actually the highest in 1988, perhaps because of the operations at a small livestock farm next to the wetland. However, after declining from 1988 to 1990, SRP and TP concentrations were substantially higher in 1993 and 1995. One of the two highest chlorophyll a concentrations in the first two years of the program was recorded at ELS61 (Reinelt and Horner 1990).

NFIC12, the wetland that had the greatest amount of development in its watershed between 1989 and 1995, increasing from 0 to 100%, displayed different water quality patterns than ELS61 and JC28. Average values for TSS rose modestly from 1989 to 1995, with a maximum peak value of 16 mg/L in 1993. Average concentrations of NH_3-N and NO_3+NO_2-N did not appear to rise during this period, but NH_3-N and NO_3+NO_2-N did reach maximum concentrations 120 and 1400 μg/L, respectively, in 1993. Concentrations of SRP and TP, however, rose steadily, reaching median concentrations of 148 and 202 μg/L, respectively, in 1995. For all years, mean and median TP concentrations exceeded 50 μg/L.

Results were less conclusive for the other two treatment wetlands, PC12 and BBC24, demonstrating that there is not necessarily a link between development and water quality degradation, even for wetlands whose watersheds have undergone similar amounts of development. A possible explanation for the difference in results may be that the watersheds of PC12 and BBC24 remained approximately half forested, retarding transport of pollutants to the wetlands in runoff. The watersheds of JC28, ELS61, and NFIC12, on the other hand were only 0 to 19.8% forested by area. In addition, a large wet pond meeting current design standards was constructed to treat storm runoff from the development built adjacent to PC12. These observations are signs that concerted action to maintain forest cover and impose structural storm water management measures can avoid water quality degradation.

Increases in nutrient loadings can have serious consequences for normally nutrient-limited bogs and fens. In one of the bog-like wetlands covered by the program in a special study, East Lake Sammamish 34 (ELS34), also known as Queen's Bog, the *Sphagnum* mat was observed to be decomposing, probably because of stormwater inflow. Nitrogen input exponentially increases decomposition rates.

Profile of Treatment Wetlands

Table 9-2 shows statistics for the five treatment wetlands in the baseline period, when little or no urbanization had occurred (1988-1990), and then the later years (1993 and 1995), after most of the changes in land use were either well underway or complete. Very little change in pH was evident. DO exhibited some fluctuation in three wetlands, but only ELS61 registered a notable decline in the median level with time (≈ 2 mg/L).

Most direct comparisons for all of the other water quality variables and all wetlands indicated no change or reduction during the program, but there were some exceptions to that generality that bear examination. NH_3-N appeared to rise in ELS61 from predominantly < to > 50 µg/L values. NO_3+NO_2-N showed increases in all but NFIC12. Median concentrations still stayed mostly < 100 µg/L in PC12 and ELS61. The increase in BBC24 kept the median still below 500 µg/L. JC28 increased from an already relatively high median > 500 to > 1000 µg/L. In NFIC12 relatively high concentrations of SRP and TP increased further after development, reaching among the highest levels seen in the entire program. Increases in TP also occurred in JC28, but stayed in the 20-50 µg/L range, and in ELS61, where the median moved from the area of 50-100 to > 100 µg/L. Relatively small rises in fecal coliform statistics were registered in JC28 and ELS61, but medians remained below 50 CFU/100 mL. Although relatively high detection limits in the early years make comparisons more difficult for the metals, there was no sign that any of the three metals increased substantially anywhere or threatened a violation of the water quality criteria applied to other water bodies.

Wetlands in moderately and highly urbanized watersheds are generally profiled earlier in this chapter. Whether or not the treatment wetlands fit these profiles after going through development will now be examined. The four M wetlands all fit the profile for that category in the cases of conductivity, NH_3-N, zinc, and fecal coliforms. It should be noted that they almost always fit the same profile in the baseline years; thus, preexisting factors are most responsible for how these wetlands profile. ELS61 and JC28 failed to fit the profile for TP and NO_3+NO_2-N, respectively, being higher in both cases. In consequence, ELS61 did not appear to be generally phosphorus-limited in photosynthetic production, in contrast to the profile. A lack of fit occurred in only one other instance, TSS in BBC24, but the median was lower than the profile value. The only highly urbanized treatment wetland, NFIC12, exhibited fewer fits to the general H profile, but usually because it had lower values. This was the case for conductivity, NO_3+NO_2-N, and fecal coliforms. It did fit for NH_3-N, TP, and Zn, but actually fit in those cases before development too. This wetland also appears to tend toward nitrogen rather than phosphorus limitation, unlike the profile. Finally, it demonstrated no tendency toward more neutral pH, as the profile states. The humic acid-producing vegetation and peat prominent in this wetland apparently were not affected by the extensive urbanization, at least not yet.

Table 9-2. Water quality statistics for treatment wetlands in baseline and post-development years.

Site/ Years	Statistic	pH	DO (mg/L)	Cond. (µS/cm)	TSS (mg/L)	NH3-N (µg/L)	NO3+NO2-N (µg/L)	SRP (µg/L)	TP (µg/L)	FC (CFU/100ml)	Cu (µg/L)	Pb (µg/L)	Zn (µg/L)
JC28 88-90	Mean	6.59	6.9	99	< 5.68	< 72	710	17	44	< 237	< 5.0	< 5.5	< 28.9
	St. Dev.	0.24	1.4	28	> 9.30	> 159	414	29	45	> 578	> 0.0	> 1.1	> 37.2
	CV	4%	21%	28%	164%	220%	58%	174%	101%	244%	0%	19%	129%
	Median	6.67	7.1	94	2.9	13	653	4	29	20	< 5.0	< 5.0	< 20.0
	n	19.00	19.0	16	19.0	19	19	19	19	19	8	8	8
93-95	Mean	6.74	6.9	98	< 4.9	< 34	1002	< 27	84	83	< 0.7	< 1.3	< 8.7
	St. Dev.	0.20	1.4	9	> 3.8	> 37	448	> 37	90	102	> 0.3	> 1.0	> 8.5
	CV	3%	20%	10%	78%	109%	45%	136%	107%	123%	45%	75%	98%
	Median	6.77	7.0	95	3.6	20	1080	8	43	36	0.6	< 1.0	< 5.0
	n	6.00	14.0	14	13.0	14	14	12	14	14	6	14	14
PC12 88-90	Mean	6.72	7.0	68	< 3.0	< 75	< 456	11	52	< 63	< 5.0	< 5.0	< 15.2
	St. Dev.	0.32	3.4	11	> 2.6	> 76	> 551	10	45	> 146	> 0.0	> 0.0	> 5.4
	CV	5%	48%	15%	88%	101%	121%	89%	87%	233%	0%	0%	36%
	Median	6.62	7.5	71	2.4	35	108	7	40	8	< 5.0	< 5.0	16.0
	n	23.00	22.0	20	23.0	23	22	23	23	23	9	9	9
93-95	Mean	6.55	6.5	73	< 2.5	< 33	< 786	< 11	< 88	46	< 0.7	< 0.8	< 3.0
	St. Dev.	0.12	3.1	15	> 2.1	> 26	> 980	> 9	> 218	124	> 0.3	> 0.4	> 1.7
	CV	2%	48%	20%	87%	79%	125%	79%	248%	271%	40%	47%	57%
	Median	6.57	6.3	75	2.0	20	430	8	24	2	0.7	0.8	2.5
	n	8.00	16.0	16	16.0	15	13	15	15	16	8	16	16
ELS61 88-90	Mean	6.59	5.3	101	13.9	< 43	< 725	< 76	166	< 100	< 5.8	< 5.1	< 17.2
	St. Dev.	0.27	3.1	19	16.8	> 94	> 1086	> 85	125	> 188	> 1.9	> 0.3	> 6.6
	CV	4%	58%	19%	121%	218%	150%	112%	76%	188%	32%	6%	38%
	Median	6.61	4.9	103	8.0	25	109	58	149	20	5.0	5.0	20.0
	n	23.00	23.0	20	23.0	23	17	22	23	23	10	10	10
93-95	Mean	6.31	< 3.8	91	< 9.5	< 136	< 527	35	101	321	< 0.9	< 0.8	< 2.3
	St. Dev.	0.19	> 2.8	19	> 20.9	> 190	> 592	41	95	992	> 0.3	> 0.3	> 1.2
	CV	3%	73%	21%	219%	140%	112%	116%	94%	309%	30%	34%	52%
	Median	6.28	3.4	90	3.0	74	344	21	62	39	< 0.9	< 0.8	< 2.5
	n	8.00	13.0	16	16.0	15	9	16	16	16	8	16	16

Table 9-2 continued. Water quality statistics for treatment wetlands in baseline and post-development years.

Site/ Years	Statistic	pH		DO (mg/L)	Cond. (µS/cm)		TSS (mg/L)		NH3-N (µg/L)		NO3+NO2-N (µg/L)		SRP (µg/L)		TP (µg/L)		FC (CFU/100ml)		Cu (µg/L)		Pb (µg/L)		Zn (µg/L)
BBC24	Mean	6.76		6.1	83	<	2.0	<	44		210		5		23	<	186	<	5.0	<	5.0	<	15.1
88-90	St. Dev.	0.25		1.9	19	>	2.0	>	35		183		3		13	>	542	>	0.0	>	0.0	>	5.6
	CV	4%		31%	23%		97%		80%		87%		64%		55%		292%		0%		0%		37%
	Median	6.77		5.4	84		1.1		31		189		4		21		14	<	5.0	<	5.0		17.0
	n	22.00		23.0	20		23.0		23		23		23		23		23		10		10		17
93-95	Mean	6.84		6.7	90	<	3.5	<	34	<	396	<	6	<	27	<	411	<	0.5	<	0.8	<	1.8
	St. Dev.	0.22		2.0	38	>	6.6	>	20	>	347	>	3	>	12	>	1492	>	0.0	>	0.4	>	1.3
	CV	3%		29%	43%		187%		57%		88%		51%		44%		362%		4%		50%		72%
	Median	6.82		7.5	82		1.7		30		323		5		28		18		0.5		0.8		2.5
	n	8.00		16.0	16		16.0		16		15		13		15		16		8		16		15
NFIC	Mean	5.08		3.4	50	<	2.3	<	41	<	54		75		119	<	2	<	5.0	<	5.0	<	19.0
88-90	St. Dev.	0.69		1.0	46	>	2.3	>	29	>	45		104		94	>	0	>	0	>	0	>	7
	CV	14%		29%	90%		98%		71%		83%		140%		79%		21%		0%		0%		37%
	Median	4.84		3.5	37		1.0		39		34		53		80		2	<	5.0	<	5.0		22.0
	n	12.00		12.0	10		12.0		12		10		12		12		12		5		5		5
93-95	Mean	4.72	<	3.6	43	<	4.0	<	40	<	477		126		253		8		2.9	<	2.2		19.6
	St. Dev.	0.18	>	2.3	13	>	5.1	>	41	>	799		95		303		16		1.1	>	1.7		9.6
	CV	4%		63%	31%		129%		102%		167%		76%		120%		207%		37%		78%		49%
	Median	4.74		3.2	39		2.0		20		20		115		177		2		2.9		1.9		18.9
	n	4.00		9.0	10		10.0		10		3		10		10		10		4		10		10

THE EFFECTS OF WATERSHED DEVELOPMENT ON SOILS

This section first profiles the soils of the urban control wetlands, both moderately and highly urbanized, using the statistical summary data in Chapter 2, Tables 2-5 and 2-6. Following the profiles is a more general summary of other applicable findings from the research.

Urbanized Wetland Soil Profiles

A soils portrait of Puget Sound Basin lowland palustrine wetlands moderately affected by urbanization shows a somewhat acidic condition, more so (by about 1 pH unit) in open water than flow-through wetlands. The range of median values can be expected to be approximately 5.1-6.1. These soils will be aerobic in many instances, but their redox potentials not infrequently are below the levels where oxygen is depleted. TP is likely to be in the range 500-1000 mg/kg, and TKN up to 10 times as high. Median levels of soil organic content are approximately 15%. No general statement is possible concerning particle size distribution. Metals appear to be less variable than in nonurban sites, but still have coefficients of variation ranging from about 60 to 100%. It is most likely for Cu concentration to be in the vicinity of 15 mg/kg, for Pb and Zn to be very roughly twice as high, and for As to be about half as concentrated. Copper and lead lowest effect threshold freshwater sediment criteria would be violated in some samples.

Only two sites represent the highly urban control sites, which is a very small sample from which to construct a profile. This group appears to have much less acidic soils than the other two, with median pH of 6.5. Soils in this urbanization category are the most likely to be anaerobic, with median redox less than 100 mv. Nutrients are no higher than in the moderately urbanized `wetlands, and may even be a bit lower. Median organic content in the small sample suggests a level of about 20%. Again, PSD is a function of local factors. The available values show metals to be distinctly higher than in the soils of the other urbanization categories, about double the values given in the preceding paragraph. These concentrations would routinely exceed lowest effect thresholds for Cu and Pb, but not for Zn. Severe effect thresholds would still not be approached often.

Other Findings

Before regular sampling began, the research program conducted a synoptic survey of 73 wetlands, about 60% urban and the balance nonurban. Samples were analyzed in the laboratory for 31 of the wetlands. In the data from this study, significant differences appeared in soil Pb concentrations between urban and nonurban wetlands in the inlet and emergent zones (Horner et al. 1988). There were also significant differences at α = 0.10 between the concentrations of both Cd and Zn in the emergent zones of urban and nonurban wetlands.

Metals accumulations may be linked to soil toxicity, as estimated by the Microtox method. The Microtox test assesses the potential toxicity of an environmental sample by measuring the reduction of the light output of bioluminescent bacteria when exposed to the sample for a period of time. The method yields the effective concentration (EC), which indicates the reduction of light output after a certain length of exposure. The lower the EC value, the more toxic the sample is (Horner et al. 1988). In the synoptic study, urban open water zone soils had significantly lower ECs in both 5- and 15-minute tests and were, therefore, relatively more toxic. There was also a significant difference in emergent zone in the 15-minute test. However, there were no significant differences between the inlet and scrub-shrub zones of urban and nonurban wetlands.

Microtox analysis of wetland soils in 1993 failed to confirm the conclusion of Horner et al. (1988) that urban wetland soils were more toxic. It should be noted that only one 1993 sample from each wetland underwent Microtox analysis, and there was no attempt to compare the toxicity of various wetland zones, as in the synoptic study. Nevertheless, the 1993 results generally indicated that urban wetland soils were certainly no more toxic than those of rural wetlands. In fact, the three soils with the most toxic compounds came from less urbanized wetlands. The extraction and concentration of naturally occurring organic soil compounds in the laboratory, and not the presence of anthropogenic toxic substances, probably explained the results for these wetlands (Houck 1994). The results suggested that the soils of FC1, an urban wetland, and AL3, a rural wetland, possibly contained anthropogenic toxicants, because the results indicated toxicity in the absence of visible organic material. There were no evident accumulations of toxicants in the AL3 soil in 1993. The FC1 wetland, on the other hand, had the highest result for Cu (59 mg/kg), a Pb concentration (60 mg/kg) second only to the highly urban B3I, and a total petroleum hydrocarbon concentration (TPH) (840 mg/kg) more than three times greater than for any wetland except B3I, which exhibited an equally high value. That metals and TPH should be high at FC1 and B3I is not surprising in view of the intensity of commercial and transportation land uses in their watersheds.

Working with 1993 data, Valentine (1994) studied the efficacy of using regression relationships between widely distributed crustal metals (aluminum, Al, and lithium, Li) and toxic metals in relatively unimpacted wetlands to evaluate whether particular wetlands have enriched concentrations of toxic metals in their soils. The method was applied independently to the 1995 data. The regression analysis is based initially on relationships between crustal and heavy metals in relatively pristine wetlands that, it is assumed, have not received significant metal loadings of anthropogenic origin. These regressions must be developed for each region, since the natural background of metals varies with soils. If, in a given wetland, the concentration of a toxic metal is above a given confidence limit (95% in Valentine's study) of the linear regression, it is probable that there has been anthropogenic toxic metal pollution of the wetland's soil.

For the purpose of her 1994 study, Valentine divided the program wetlands into the same three groups outlined earlier in this chapter. She used the nomenclature Group 1 for nonurban (N) wetlands, Group 2 for moderately urbanized (M) ones, and Group 3 for highly urbanized (H) cases. In 1996, she employed only two groups, one less urban and the other more urban. Using the 1993 soil metals data, Valentine (1994) found that Li may be as good or better a reference metal than Al for As, Pb, and Zn. Nickel (Ni) bore a stronger relationship to Al, while Cu correlated equally well with both Al and Li.

Figures 9-1 and 9-2 illustrate the assessment tool using a nickel-aluminum pairing with 1993 and 1995 data, respectively. The regression line represents the best-fit line of the Group 1 (N) wetlands. The 95% confidence limits are the upper and lower bounds for one additional sample that is being assessed for Ni contamination. Sample contamination is gauged by considering the corresponding point’s location on the graph. If the point lies on or above the upper 95% confidence limit, then the sample is judged to be enriched with the contaminating metal. Thus, in Figure 9-2, for example, seven samples above the line were judged to have Ni contamination of anthropogenic origin. The two relationships from the separate data sets exhibit a close correspondence.

Valentine (1994) found that the most urbanized wetlands had a higher rate of soil metals enrichment than moderately urbanized wetlands, considering both the Al and Li regressions. There were far fewer indications of metals enrichment in the moderately urbanized wetlands in 1993. The regressions of 1993 Pb with both Al and Li strongly agreed that most soil samples from each of the most urbanized wetlands were Pb enriched. The first set of regressions of As with both Al and Li generally agreed that soil samples from each of the most urbanized wetlands were As-enriched. Both Cu regressions using 1993 data indicated Cu enrichment in two of the most urbanized wetlands, which are also the wetlands listed as highly urbanized wetlands in Table 1 of Section 1. However the Li-Cu regression using 1995 data indicated enrichment in only two nonurban and one urban wetlands. The Al-Ni regression with 1993 data indicated Ni enrichment in three of the four most urban wetlands, although the Li-Ni relationship failed to show any enrichment in these sites. For 1995 data, no Ni enrichment appeared in the less urban wetlands, while there were five and six cases of enrichment according to the Al-Ni and Li-Ni regressions, respectively, in the more urbanized wetlands. The Li-Ni regression using 1995 data showed enrichment in all cases in which the Al-Ni regression also indicated enrichment. The first set of regressions for Zn showed a few cases of enrichment in the most urban group, although fewer in number and with less agreement between the regressions than for the other metals. The Li-Zn relationship in the 1995 data indicated Zn enrichment in ten of the more highly urbanized wetlands, in comparison to only one of the less urbanized wetlands.

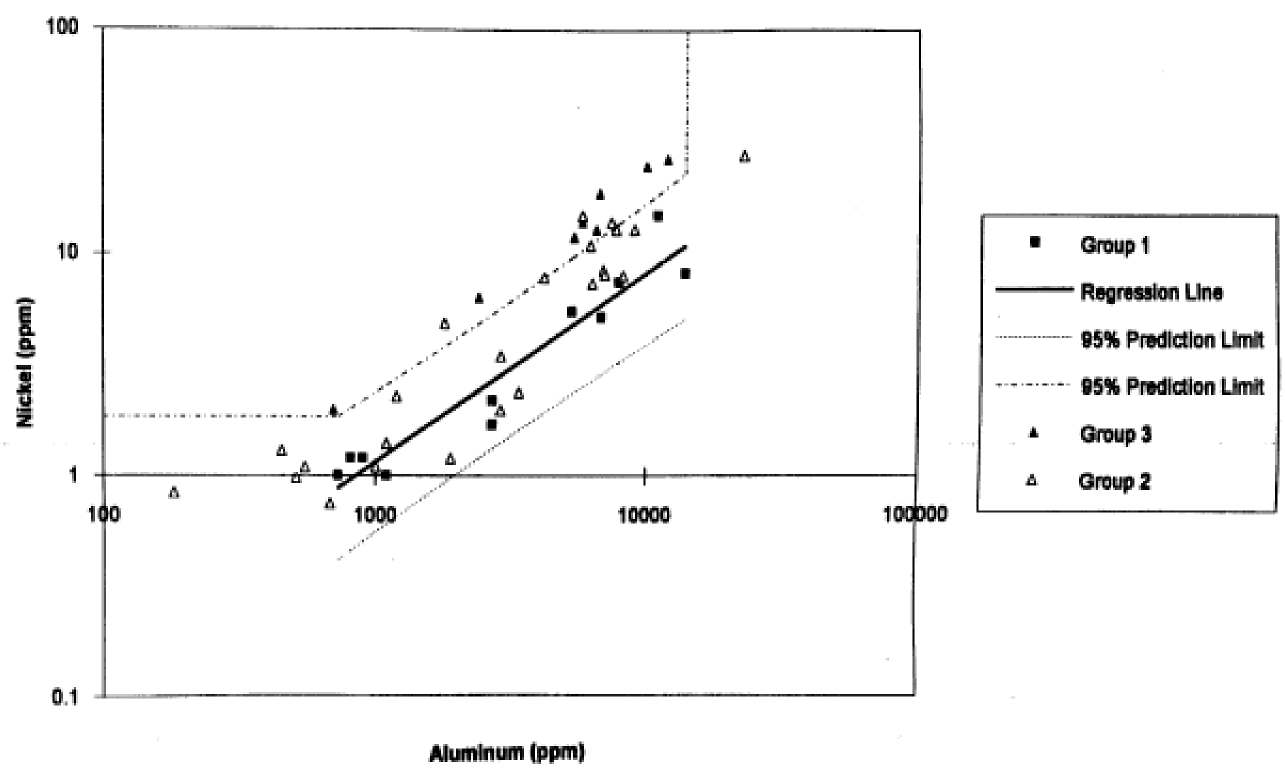

Figure 9-1. The Assessment Tool for NI Using AI as the Reference Element 1993 Data.

Although Valentine classified some of the wetlands in different groups than they would be in according to the GIS analysis, the results of her study agree well with observations based on the soil data statistics. Therefore, anthropogenic sources clearly impact the sediments of palustrine wetlands in the Puget Sound Basin. While wetlands can remove metals from the water column, the accumulation of metals could still harm wetland functions. Valentine noted that long-term effects of atmospheric emissions from past operations of the ASARCO smelter in Tacoma on wetland soils are unknown. It is possible that such distant sources could play a role in the enrichment of toxic metals wetlands. Rainfall removes such suspended metals from the atmosphere and provides the runoff which transports the metals to wetlands, where they accumulate in the sediments.

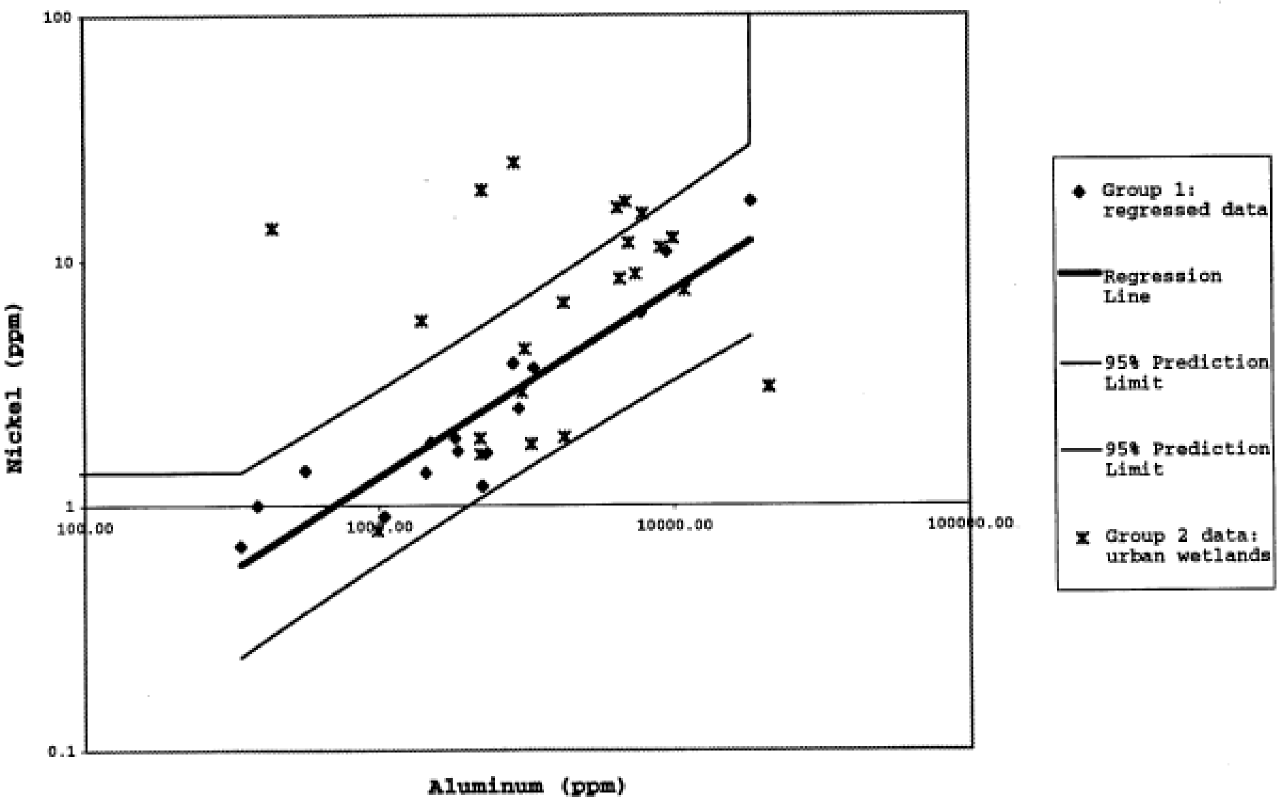

Figure 9-2. The Assessment Tool for NI Using AI as the Reference Element (1995 Data).

Treatment Wetlands

Table 9-3 shows soil statistics for the five treatment wetlands in the baseline period, when little or no urbanization had occurred (1988-1990), and then the later years (1993 and 1995), after most of the changes in land use were either well underway or complete.

It appears from program data that soil pH may have increased over the years at BBC24 and, especially, NFIC12, the wetlands whose watersheds underwent the greatest amounts of development. For NFIC12, rises in pH are entirely expected because (1) this wetland was a late successional peat bog that had the lowest soil pH readings of any of the wetlands, with no place to go except up; and (2) its watershed went from 0% to 100% urbanization between 1989 and 1995.

The treatment wetlands exhibited the particle size distributions in 1989 and 1995 shown in Table 9-4. BBC24 and ELS61 both registered transition from relatively sandy to silty soils, while clay stayed constant. NFIC12 exhibited a clay increase while sand was constant. The first result could be explained by sedimentation of finer particles over the pre-development substrate, but a 30% increase in clay is not easily explained.

Table 9-3. Soil statistics for treatment wetlands in baseline and post-development years.

Year	Stat.	pH	Redox. Potential (mV)	TP (mg/kg)	TN (mg/kg)	Volatile Solids (%)	Cu (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	As (mg/kg)
JC28	Mean	5.89	12	964	6661	49.6	19.5	33.2	27.1	6.4
88-90	St. Dev.	0.11	240	637	4578	26.5	3.2	15.1	23.7	2.0
	CV	2%	2003%	66%	69%	53%	16%	46%	88%	31%
	Median	5.87	-20	578	6056	59.9	20.4	32.6	18.0	7.1
	n	7.00	7	7	8	9.0	7.0	7.0	7.0	7.0
93-95	Mean	5.63	545	153	1392	13.1	5.0	8.1	3.4	1.8
	St. Dev.	0.32	102	150	1241	1.2	2.6	2.3	2.4	1.6
	CV	6%	19%	98%	89%	9%	51%	28%	71%	89%
	Median	5.53	608	176	1980	12.8	4.3	8.5	2.7	1.2
	n	5.00	5	5	5	5.0	5.0	5.0	5.0	5.0
ELS61	Mean	5.70	162	755	4409	33.6	31.8	60.3	47.8	9.0
88-90	St. Dev.	0.34	187	716	3821	22.6	20.3	20.3	24.8	5.4
	CV	6%	115%	95%	87%	67%	64%	34%	52%	60%
	Median	5.84	70	1024	3443	29.5	31.0	32.4	56.6	9.2
	n	11.00	11	13	14	14.0	13.0	13.0	13.0	13.0
93-95	Mean	5.70	164	748	4382	33.8	31.6	60.2	47.6	9.0
	St. Dev.	0.71	298	973	5092	29.3	158.4	77.9	64.1	16.8
	CV	13%	182%	130%	116%	87%	502%	129%	135%	187%
	Median	5.67	547	69	537	8.1	12.7	9.5	14.3	2.8
	n	191.00	188	194	204	208.0	195.0	195.0	195.0	195.0
PC12	Mean	6.08	107	1285	8431	52.5	26.1	170.7	52.7	14.3
88-90	St. Dev.	0.29	291	1122	6813	23.2	8.7	197.8	31.5	7.2
	CV	5%	273%	87%	81%	44%	33%	116%	60%	50%
	Median	6.04	114	1089	6347	59.2	26.9	105.1	39.7	12.5
	n	7.00	7	6	7	6.0	6.0	6.0	6.0	6.0
93-95	Mean	6.15	351	111	911	6.9	3.7	9.1	7.5	1.7
	St. Dev.	0.14	99	115	866	1.5	0.9	6.0	3.2	0.5
	CV	2%	28%	104%	95%	21%	24%	65%	43%	32%
	Median	6.10	310	109	1060	7.0	3.6	10.3	6.6	1.8
	n	5.00	5	5	5	5.0	5.0	5.0	5.0	5.0
BBC24	Mean	5.90	-197	469	4171	22.7	19.1	44.9	39.0	11.0
88-90	St. Dev.	0.35	15	434	4802	17.6	12.4	25.2	16.5	5.5
	CV	6%	-8%	93%	115%	77%	65%	56%	42%	50%
	Median	5.73	-200	297	1692	15.3	17.9	51.6	48.0	10.7
	n	6.00	6	6	7	7.0	7.0	6.0	6.0	7.0
93-95	Mean	5.97	326	43	441	7.4	5.0	4.4	6.0	1.4
	St. Dev.	0.24	49	54	596	3.7	2.9	0.7	0.5	0.3
	CV	4%	15%	126%	135%	50%	58%	16%	9%	26%
	Median	5.97	326	43	441	7.4	5.0	4.4	6.0	1.4
	n	2.00	2	2	2	2.0	2.0	2.0	2.0	2.0

Table 9-3 continued. Soil statistics for treatment wetlands in baseline and post-development years.

Year	Stat.	pH	Redox. Potential (mV)	TP (mg/kg)	TN (mg/kg)	Volatile Solids (%)	Cu (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	As (mg/kg)
NFIC12	Mean	3.97	144	2188	15984	76.0	19.5	48.6	10.4	9.2
88-90	St. Dev.	0.26	345	1138	7440	18.9	10.1	40.0	5.3	5.6
	CV	7%	240%	52%	47%	25%	52%	82%	51%	61%
	Median	3.94	140	2182	15698	81.0	16.3	44.4	10.9	9.9
	n	4.00	4	4	4	4.0	4.0	4.0	4.0	4.0
93-95	Mean	5.05	633	167	1950	12.5	2.5	10.3	3.0	2.2
	St. Dev.	1.05	2	67	113	0.7	0.3	0.8	2.2	0.3
	CV	21%	0%	40%	6%	6%	13%	8%	71%	13%
	Median	5.05	633	167	1950	12.5	2.5	10.3	3.0	2.2
	n	2.00	2	2	2	2.0	2.0	2.0	2.0	2.0

Table 9-4. Particle size distributions for treatment wetlands.

	%Sand/Silt/Clay	
Wetland	1989	1995
BBC24	61/29/10	45/47/8
ELS61	35/49/16	15/69/16
JC28	52/35/13	46/36/18
NFIC12	4/59/37	3/32/65
PC12	58/37/5	63/30/7

Otherwise, there was a strong trend for redox to rise but for nutrients, organic content, and metals all to fall from the pre-development to the post-development years. The reasons for these unexpected results can only be given speculation. What can be said is that, other than the pH increase at NFIC12, there is no obvious signal in the soils yet that negative changes may have accompanied recent urbanization.

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