Section 7.0
SOILS AND SEDIMENTS AS INDICATORS OF AGRICULTURAL IMPACTS ON NORTHERN PRAIRIE WETLANDS

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7.1 INTRODUCTION

The northern prairie wetlands are, for the most part, the products of glaciation that ended less than 13,000 years ago (Bluemle 1991). Landscapes pocked by these wetlands are characterized by internal drainage. Watersheds are the surrounding drainage basins that contribute runoff, ground water seepage, dissolved solids, and eroded sediments to the wetlands located at the bottom. Some watersheds are hydrologically isolated, while others are hydrologically and geochemically connected to others by runoff or groundwater flow. Land use practices, especially agriculture, may have a significant impact on both the quality and quantity of materials that enter wetland communities.

In time, wetland habitat will be lost or seriously damaged due to one or a combination of the following natural or anthropogenic processes: (1) contamination by toxic levels of salts and other chemicals, (2) establishment of an integrated stream drainage system, (3) long-term drought associated with subsequent lowering of water tables, or (4) filling-in by inorganic sediments and organic matter. While PO_3 is readily adsorbed, NO_3 is more likely to be transported in soluble form. Sediments often carry phosphate and nitrate fertilizers (Neely and Baker 1989), which may enhance plant growth and hasten the accumulation of organic sediments within the wetland basin. Elevated phosphorous levels can promote eutrophication in aquatic ecosystems and enhance the growth of blue-green algae (Schindler 1977, Crumpton 1989). Algae can become a problem to the point of causing fishkills in prairie pothole lakes (Kling 1975). Chemical sediments, especially salts, are common in the subhumid and semiarid prairie potholes. Salts can severely limit the condition and productivity of the potholes (Richardson and Arndt 1989). However, some wetlands in the PPR contain plants and animals well adapted for normal salinity, but elevated salinity from certain land-use practices are problems.

Soils are an essential part of the wetland ecosystem, serving as both a reservoir of water and nutrients and as a medium for biogeochemical processes. To conserve wetland ecosystems, we need to understand processes that threaten them, be able to measure those processes and develop acceptable protective strategies. Our primary objective in this study was to evaluate, within the constraints of the project, the extent to which soils reflect the impact of land use, and identify contrasting soil indicators that distinguish good-condition from poor-condition wetlands (See Section 2 for condition definition). To assess and monitor the condition of these wetlands, one must have a concept of what constitutes good-condition wetlands, and be able to measure the processes which lead to their demise. We are proceeding with the assumption that good-condition wetlands are those closest to being pristine, i.e. undisturbed by human intervention, and poor-condition wetlands are those most disturbed by human practices, especially agricultural tillage and cropping.

7.2 OBJECTIVES

1. Determine if good- and poor-condition wetlands can be distinguished from each other.
2. Determine the quality and quantity of sediment entering wetlands seasonally as a result of erosion.
3. Determine the long-term sedimentation rates in good- and poor-condition wetland landscapes.
4. Identify and measure some key soil constituents that reflect land use impacts, and pose a threat to the condition of wetland ecosystems.

7.3 METHODS

7.3.1 Quality and Quantity of Sediments

Sediment trapping devices (see Section 9-1) were installed in 35 sample wetland basins. The samples from the traps were analyzed for invertebrate remains at the Northern Prairie Science Center (NPSC) as described in Section 5.6.1. The frozen sediment slurries were then mailed to us at North Dakota State University (NDSU). We stored the samples for about one week in a walk-in refrigerator maintained at a constant 3 °C. While in the refrigerator, the samples melted and the sediment settled to the bottoms of their plastic containers. After settling, the clear water was decanted off and the remaining sediment samples were dried for 24 hours in a forced-air evaporating oven at 65 °C. Since many of the samples from individual traps were too sparse for analysis, we composited the sediments from each wetland into a single sample. Analyses included

- organic matter content using the loss-on-ignition method (Schulte 1988)
- calcium carbonate ($CaCO_3$) equivalent by the Williams (1948) method
- phosphorus by the sodium bicarbonate ($NaHCO_3$) extraction method (Olsen et al. 1954, Knudsen and Beegle 1988).

We had planned to perform particle size analysis on the sediment samples, however, sediment amounts were often too small to conduct the analysis.

7.3.2 Long-term Sedimentation (Cottonwood Lake Study Area)

Cesium-137 (Cs-137) is a radioactive isotope introduced to the atmosphere in the 1950's by way of atomic weapons testing. Maximum atmospheric levels of Cesium-137 were detected in 1954, and the winter of 1963-64. Cesium is tightly adsorbed to sediment particles and serves as a marker for estimating sedimentation rates since 1954.

Several studies (DeLaune et al. 1978) have used peak Cs-137 levels found in sediment profiles to successfully establish time markers and interpret depositional histories. The dating of vertical sediment accumulation using Cs-137 peaks in sediment profiles depends on a major assumption we feel we can not make in the northern prairie pothole wetlands. To use the profile-peak method we must assume a constant sedimentation rate (Ritchie et al. 1973). However, using a hypothetical example, suppose fallout Cs-137 entered a wetland basin in 1954 and remained attached to upland soils for many years of relative drought and idle land use. If the basin was later disturbed by cultivation, then received heavy precipitation in 1962, causing the land to erode, sediment laden with Cs-137 would enter the wetland basin in 1962. If a researcher using the profile-peak method (described above) sampled those new wetland sediments, which were high in Cs-137, they would probably interpret those sediments as having been deposited in 1954, the first peak year for Cs-137 fallout. This, of course, would be an erroneous interpretation, since the sediments actually were deposited in 1962. Considering the highly variable climate here and the alternating drawdown and emergent phases of northern prairie wetlands, constant sedimentation rates seem highly unlikely. Additional problems with the profile-peak, in our opinion, exist. For example, recharge events following a severe drought in a dry wetland may cause clay particles bearing Cs-137 to be leached to lower profile depths. Cattle and other animals common to the northern prairie wetlands may distort Cs-137 horizons by disturbing and mixing wetland sediments.

For these reasons, we used another method similar to that used by Soileau et al. (1990), and DeJong et al. (1986) to estimate average annual soil erosion rates. The method does not assume constant sedimentation rates and focuses on soil loss occurring from 1954, the year when Cs-137 was introduced to the atmosphere by weapons testing. The soil loss rate is averaged over 39 years. It makes no difference how sporadic were the periods of erosion and deposition. Although we focus on erosion using this model, not sedimentation, we think it is safe to expect watershed erosion to be directly related to wetland basin sedimentation. Furthermore, it seems reasonable to expect that any future land management practices aimed at reducing wetland sedimentation will have to directly address watershed erosion.

We used the following analog of Soileau et al. (1990) to estimate annual erosion in cultivated and uncultivated wetland watersheds, where:

$A = [(B-C)/B] * D/E$

A = annual rate of soil erosion (metric tons/ha),

B = total Cs-137 activity (Bq/m^2) in 0-15 cm cores of baseline flat, non-eroded site,

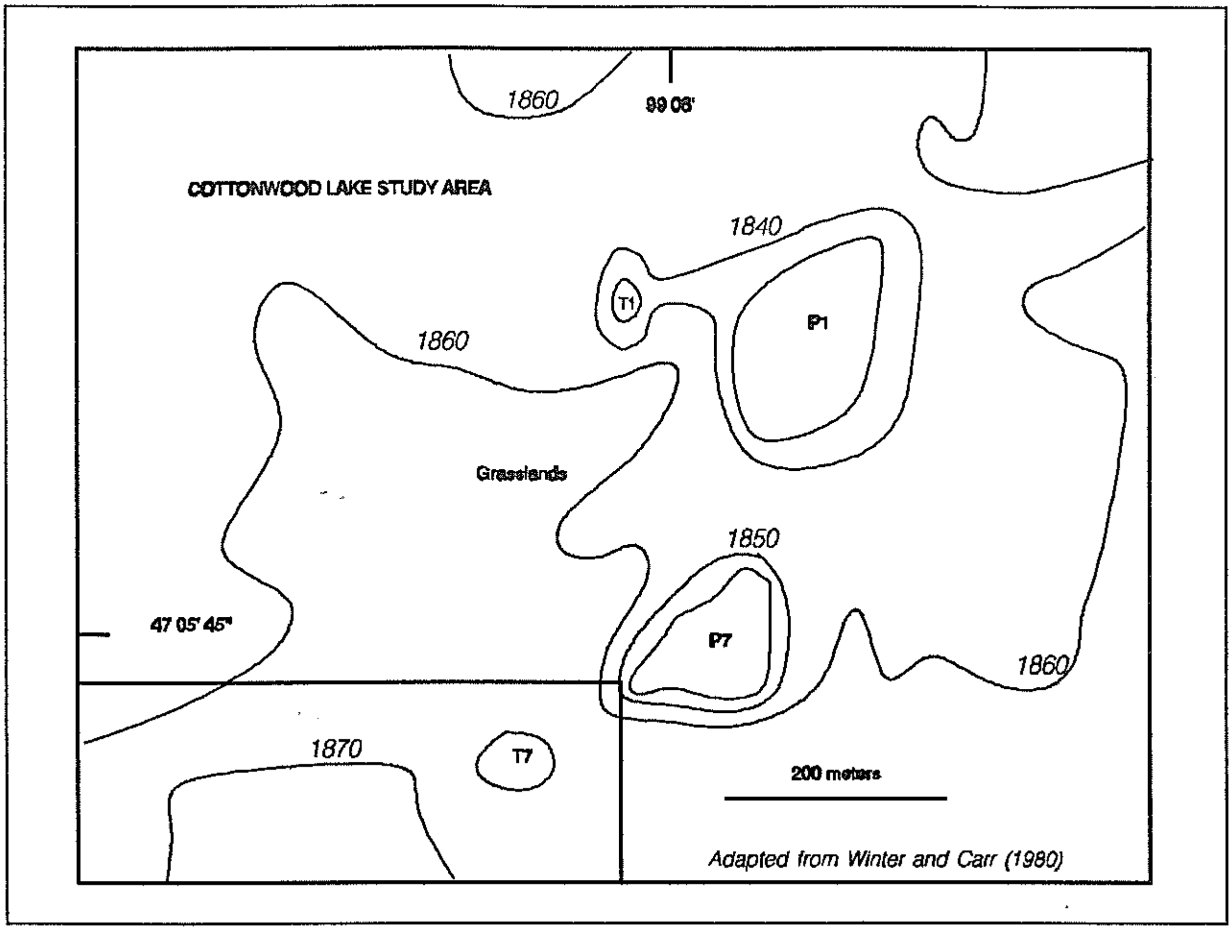
C = total Cs-137 activity (Bq/m^2) in 0-15 cm soil of eroded side slopes.

D = soil mass in 0-15 cm depth core (Mg/M^3) * 1 ha volume of soil (metric tons),

E = years elapsed between initial Cs-137 fallout and soil sampling (39 years for this study).

Because of the high cost of Cs-137 analyses, we tested this method on four distinct sites as a preliminary investigation of using Cs-137 to analyze sedimentation.

To determine long-term sedimentation rates in wetlands surrounded by cultivated versus uncultivated fields, we collected soil samples from side slopes of four wetland watersheds (P1, P7, T1, and an unnamed wetland basin on private property designated C7 for this study)in May 1993 (Fig. 7-1). P1 and P7 are semipermanent wetlands; T1 and C7 are seasonal wetlands (Stewart and Kantrud, 1971). P1 and T1 are surrounded by grassland, C7 and part of P7 are surrounded by cultivated fields.



Adapted from Winter and Carr (1980)

Figure 7-1. Map of the Cottonwood Lake study area.

We collected duplicate samples, 5 cm in diameter by 15 cm deep, using a slide-hammer coring device from three different points, all on side slopes, surrounding each of the four basins. Three samples of the same dimensions were collected from a flat, uneroded site northwest of P7 to use for control (factor B in the soil loss equation).

Subsamples were oven-dried at 105 °C to determine hygroscopic moisture and soil bulk density. We composited the samples from each basin into single samples. Approximately 700 g of soil composite were placed in four Marinelli beakers, one for each wetland, and analyzed twice for Cs-137 activity using gamma counting equipment. Counting time was 16 hrs. The gamma ray sensor was a 1.5 in. X 1.5 in. (3.8 cm by 3.8 cm) ORTEC 905-2 NaI Scintillation Detector coupled to a Canberra Series 85 Multichannel Analyzer. Analytical software was the MAESTRO II Emulation Software Model A64-BI Version 1.40 (EG&G ORTEC, 1991, Oak Ridge, TN).

7.3.3 Key Soil Constituents

7.3.3.1 Cottonwood Lake Study Area

Soil samples and data were collected at CWLSA in early June 1992. We collected soil samples and data from four wetlands and wetland watersheds, including P1, T1, P7, and C7 (Figure 7-1). Soil samples were collected from wetland vegetation zones (Stewart and Kantrud, 1971) along radial transects. To establish the transects, we measured salinity every 10 paces along the wet meadow zone of each of the four wetlands using the GEONICS EM-38 (Geonics LTD, Mississauga, Ontario). Other transects were placed at about 100 m intervals, as measured by pacing, along the wet meadow. P1, the largest wetland had 10 transects, P7 had 6, and T1 and C7 each had four transects. T1 and C7 are relatively small wetlands, and the transects had to be spaced closer together to have a minimum of four transects per wetland.

We collected soil samples from profiles where the transects intersected the wetland vegetation zones at four depth increments per profile (0-15 cm, 15-30 cm, 30-45 cm, and 45-60 cm). We placed about 400 grams of each sample in plastic-lined bags and stored them in coolers until late afternoon of each field day, when we returned to the Woodworth Field Station. Here the samples were spread out in a garage to air-dry. Back at NDSU, the air-dried samples were sieved through a 2-mm screen. All soil laboratory analyses were conducted on dry, sieved samples.

We classified soil profiles in the field using *Keys to Soil Taxonomy* (Soil Survey Staff 1975). Watershed measurements consisted of extending the linear transects away from the wetland to the top of the wetland watershed divide. The length of each upland transect was measured by pacing. We also measured steepness with a pocket clinometer.

The specific soil characteristics that we tested for their use as potential condition indicators included

- *Soil Classification* (Soil Survey Staff, 1975)
- *Nitrate-Nitrogen* by transnitration of salicylic acid (Vendrell and Zupacic 1990)
- *Sodium bicarbonate-extractable Phosphorus* (Knudsen and Beegle, 1988)

- *Organic Matter* by loss-on-ignition (LOI) (Schulte 1988)

- *Soluble Salts* by electrical conductivity (EC) (Dahnke and Whitney 1988)

- *In Situ Salinity* using a GEONICS EM38 electromagnetic induction meter

- *Soil pH* in 2:1 0.01M $CaCl_2$ solution-to-soil slurries (Eckert 1988)

- *Particle Size* using Particle Size Analysis (PSA) by the hydrometer method (Day 1965).

The nitrate, phosphorus, and organic matter tests were performed at the NDSU Fertility Laboratory; soluble salts, pH and PSA were performed in the NDSU Soil Characterization Laboratory.

The sodium bicarbonate ($NaHCO_3$)-extractable P is sometimes known as the "Olsen" test (Olsen et al. 1954) after its originator. This is a relatively inexpensive procedure that yields values which have correlated well with crop responses. This test is the one routinely used on agricultural soils in North Dakota in order to make fertilizer recommendations. Since we are concerned with fertilizer additions to wetlands, the Olsen test seems appropriate, since it is sensitive to common fertilizer sources of P. Wolf et al. (1985) found fairly good coefficients of determination (0.71) between Olsen test P and algal available P.

The LOI method (Schulte 1988) was developed as an alternative to the more complex and time-consuming Walkley-Black (1934) carbon test, which is a wet-chemical procedure. The LOI method correlates very strongly to the Walkley-Black test and, unlike Walkley-Black, requires no hazardous chemicals. The LOI test includes first drying a sample to 105 °C, and recording its dry weight. Next the sample is baked in a muffle furnace at 360 °C for 2 hours and weighed again. Although LOI tests are often run at 450 °C or higher; in prairie regions, 360 °C should be the standard procedure. The weight loss is due to the oxidation of organic matter (OM) in the soil.

The test for soluble salts by electrical conductivity (EC) was performed on 1:1 soil to distilled water slurries. Twenty ml of water were added to 20 g of soil, stirred, and left to stand for 15 minutes before measuring electrical conductivity with a Type 700 Conductivity Meter (Chemtrix Corp., Hillsboro, OR). Soil pH was measured using an ORION Ion Analyzer Model 901 (Cambridge, MA).

Particle size analysis was performed on samples collected from four depth increments: 0-15 cm, 15-30 cm, 30-45 cm, and 45-60 cm. Prior to analysis, samples were air-dried and sieved through a 2-mm screen. We used a hydrometer method similar to that described by Day (1965) with some modifications intended to save time and allow for the analyses of greater numbers of samples. Our method does not include digestion of organic matter, washing salts, or shaking overnight; it emphasizes chemical instead of mechanical dispersion. The procedure was as follows:

1. Weigh 40 g of soil and place it in a hydrometer jar.

2. Add 100 ml of Calgon dispersant.

3. Add enough distilled water to the Calgon to make 1 liter.

4. Agitate the suspension with 30 up-and-down plunger cycles and let sit overnight.

5. In the morning, check temperatures, agitate again with 30 plunger cycles and begin hydrometer readings.

6. Read hydrometer values at 1, 3, 10, 30, 90, 270, and 480 minutes.

7. Calculate sand, silt, and clay percentages using a LOTUS 123 spreadsheet in the Soil Characterization Laboratory.

The procedure saves time on bottle washing since there is no overnight shaking in a flask or drink mixer. Again, we performed no salt washing or OM digestion.

Multiresponse permutation procedure (MRPP) is a statistical method of analyzing ecologic data which do not necessarily conform to assumptions of normal distribution and equal variances required when using least squares analyses such as linear regression and analysis of variance (Biondini et al. 1988). Since some of the data generated by this study do not fit the normal distribution, use of MRPP seemed appropriate.

MRPP is a method based on absolute Euclidean distances (Biondini et al. 1988). Distances are calculated between all possible pairs of points in each group and averaged to calculate the group distance value. Then, the group distance values are weighted according to the number of samples in each group and averaged together to calculate the delta value. If the groups are real, i.e., they form separate groups in the data space, the delta value will be significantly low. If, however, the assigned groups are not really different, alternative groupings, or permutations, may yield distance values smaller than the values calculated for the chosen groupings. In this latter case, the P-values would be high, since the chance of getting lower delta values would be relatively high.

The basic statistical data are in Tables 7-1 and 7-2. MRPP was applied to nitrate, P, and OM data from the 0-15 cm depth samples collected from each transect. The values shown in Table 7-3 are from wet meadow soils, or in cases where wet meadow was absent, shallow marsh soils. The samples shown come from the 0-15 cm depth increment. These samples were chosen for primary analysis since they are closest to the surrounding land use activities, i.e., they are on the edge of the wetland, on or near the land surface. Samples were separated into two conditions, based on the land use adjacent to and up-slope from the sample site. In P7, transects 2 and 6 were somewhat intermediate in their adjacent land use. The immediately-surrounding land use is grassland for those two sites, but, we think, they are sufficiently close to the cultivated fields that they warrant placement in the cultivated (poor-condition) group. Seventeen samples fell into the grassland (good-condition) group, and 7 were placed in the cropland (poor-condition group).

7.3.3.2 10.4-km^2 Sample Plots

In 1992, we sampled 40 wetlands at the same sites where H. Kantrud took plant data (see Section 6.3.1) and collected soil samples from vegetation quadrats placed in delineated plant communities. In each community, soil profiles from five quadrats per community were classified, and soil was collected from quadrats two and four for laboratory analysis. Soil profiles were dug, using a "Dutch" auger, to a depth of about 75 cm, sufficiently deep to classify the soil and collect laboratory samples. Two samples were collected from each profile, one from 0-15 cm depth and another from 15-45 cm. Laboratory analyses included all those performed at the CWLSA wetlands except PSA, i.e., NO_3^- $NaHCO_3$-extractable P, organic matter, EC and pH. As in the CWLSA study, we measured basin size by pacing from the wet meadow to the basin divide along at least 4 transects. We also measured *in situ* salinity along the wet meadow using the EM-38 in the same manner described above for the CWLSA study.

In 1993, we sampled soils from 36 randomly selected wetlands according to wetland vegetation zones from H. Kantrud's study (see Section 6). We sampled and classified soil profiles from 3 of the 5 quadrats selected per community. As in 1992, we collected samples at 2 depths, 0-15 cm and 15-45 cm. Samples were bagged in plastic-lined paper bags and stored in coolers, as they were in 1992. Approximately 20 grams of each bagged sample were placed in aluminum cans, which we opened in the evening of each field day and placed in a forced-air evaporating oven. We carried the oven in our vehicle, setting it up in a motel room in the evening. Samples oven-dried at 65 °C overnight at a motel. Prompt drying is necessary to prevent analytical errors due to potential nitrogen transformation by microbes in the sample bags (Dahnke, 1988). The oven-dried samples were lightly ground, sieved through a 2-mm screen and used for NO_3^-, P and OM analyses. The bagged samples were used for analysis of pH, EC, and PSA. We only ran PSA on the 0-15 cm depth samples. Soil pH in 1993 was measured in distilled water, instead of 0.01 *M* calcium chloride ($CaCl_2$) as was used in 1992. In the above laboratory tests, except particle size analysis, every 10th sample was replicated.

We used analysis of variance (ANOVA) techniques to assess the effects of wetland condition, zone (deep marsh, shallow marsh, and wet meadow), depth, and year (1992, 1993) on the response variables NO_3, P, OM, EC, pH, sand, silt, and clay. The design was a strip-split-plot with repeated measures. Each basin was assumed to be the independent whole-unit, with zone-depth and community combination being the subunit. Because most watersheds were measured in both 1992 and 1993, year served as the repeated measures factor. We used Fisher's protected least squares differences (LSD) to isolate differences in least squares means following significant effects in the ANOVAs (Milliken and Johnson 1984). All ANOVAs were done using the general linear model procedure (PROC GLM) of SAS (SAS Inst. 1992). A 1n(y+1) transformation was done on all data except pH prior to analysis because the data were skewed to the right (note: clay was only marginally skewed). Data were averaged across quadrants (2 in 1992, 3 in 1993) prior to 1n(y+1) transforming. Sand, silt, and clay were only measured at the 0-15 cm depth in year 1993. Statistical tests were considered significant at the 0.05 level, and marginally significant at the 0.10 level.

7.3.4 Soil Oxidation-Reduction

We placed platinum electrodes in the three wetland zones (wet meadow, shallow marsh, and deep marsh) of wetlands T1, P7, and C7. We wanted to monitor oxidation-reduction potential to see if the hydric soil morphology observed in the sampled profiles at CWLSA corresponded to active oxidation-reduction processes, or, alternatively, the hydric soil morphology was possibly relict, i.e., developed during some wet climatic episode in the distant past. The electrodes were to be monitored during the frost-free months of September 1992 to November 1993.

7.4 RESULTS

7.4.1 Seasonal Sedimentation

Results of chemical analyses of trapped sediments are shown in Table 7-1. In this table, CRP land is considered good-condition because short-term sedimentation is expected to reflect the most recent activities within the drainage basin. Average amount of sediment per trap and phosphorus concentrations were nearly equal for the good- and poor-condition groups. Average organic matter in the good-condition group (15.1 g/100 g) was significantly greater (F=7.14, df 1,29 P = .012), than in the poor-condition group (9.4 g/100 g). Calcium carbonate ($CaCO_3$) equivalent (CCE) is a measure of that percentage of the sediment mass attributable to $CaCO_3$. We did not have sufficient sample to perform this test on many samples and, therefore, we are reluctant to draw any comparisons between condition groups. In sediment analysis, we expected high variation because of the wide settlement rate due to minor landscape and vegetation differences, creating, large settlement differences, both in amount and quality of sediment. Sediment will always have problems in use.

7.4.2 Long-term Sedimentation

Estimations of soil loss for the four wetland basins P1, T1, P7, C7 and a flat, noneroded site used for making comparisons are shown in Table 7-2. Using the formula from Soileau et al. (1990), we calculated soil loss (tons/ha) for each basin. As expected, the cultivated basin C7 had the greatest soil loss from its side slopes (35.6 metric tons/ha/year), and P7, the basin partially surrounded by cultivated fields had the second highest soil loss (11.71 metric tons/ha/year). T1 soil loss was 4.56 metric tons/ha/year. The P1 soil loss value (+16.13) indicates a net gain in Cs-137 and soil deposition compared to the noneroded control site.

7.4.3 CWLSA Soil Characterization

Results of laboratory analyses for fertilizer nutrients NO_3^-, P and OM are shown in Table 7-3.

Results from the MRPP (Table 7-4) show that P is the strongest contrasting variable separating the good from the poor group, while the NO_3 variable alone is the least significantly different.

Table 7-1. Analysis of selected chemical constituents of trapped sediments, 34 sample wetlands, North and South Dakota, collected in 1993. Abrev.: OM = organic matter, CCE = calcium carbonate equivalent, SD = standard deviation.

Plot	Wetland	Condition	Dry Density (g/cm^3)	P g/m^3	P g/Mg	Sediment g/trap	P loading rate g/Mg	% OM	% CCE
73	29	Good	0.39	69.6	178.5	0.87	155.3	11.9	INS
73	86	Good	0.48	35.2	72.7	1.04	75.6	23.9	0.0
133	370	Poor	0.76	55.4	73.2	3.53	258.3	8.6	0.0
133	380	Poor	0.39	111.0	285.9	0.95	271.6	INS	INS
133	386	Good	0.39	139.0	358.0	0.27	96.7	INS	INS
134	140	Poor	0.36	38.8	106.9	0.90	96.2	11.8	INS
134	158	Poor	0.78	115.6	147.8	6.06	895.4	8.1	3.8
134	165	Poor	0.55	93.8	169.5	1.58	267.7	10.5	INS
134	270	Poor	0.54	54.6	102.0	0.88	89.8	13.1	1.8
134	406	Poor	0.82	46.8	57.3	2.13	122.0	7.5	0.6
134	432	Poor	0.45	36.1	79.4	0.65	51.6	10.7	INS
156	22	Good	0.62	39.1	63.2	6.43	406.3	14.3	0.0
156	26	Good	0.68	45.5	66.9	7.89	527.9	10.0	0.0
156	42	Good	0.39	89.4	230.3	0.34	78.3	INS	INS
327	72	Poor	0.74	75.8	101.9	2.56	261.0	9.1	3.6
327	117	Poor	0.70	47.7	68.3	4.75	324.5	11.4	0.0
327	147	Poor	0.57	54.0	94.7	4.32	409.3	12.5	0.0
363	22	Good	0.47	14.6	31.0	9.31	288.5	11.4	0.3
363	58	Good	0.36	23.5	65.6	0.45	29.5	18.6	INS
374	65	Good	0.50	60.4	121.9	4.20	512.2	26.4	10.2
374	100	Good	0.32	32.5	101.6	1.40	142.2	25.3	2.4
374	225	Good	0.38	50.3	132.8	5.98	794.0	24.1	0.0
374	272	Good	0.72	13.7	19.1	1.45	27.7	3.7	4.9
407	67	Good	0.65	54.2	83.8	4.17	349.3	13.0	0.0
407	109	Good	0.43	36.6	84.7	0.43	36.4	21.0	INS
407	168	Good	0.83	43.8	53.0	9.63	510.4	6.1	10.0
442	93	Good	0.53	60.4	113.3	1.37	155.3	11.6	INS
442	260	Poor	0.88	56.2	64.2	6.12	392.7	5.7	0.0
442	261	Poor	0.86	64.0	74.6	5.23	390.0	5.9	0.0
442	281	Poor	0.66	64.0	97.0	2.62	254.1	7.7	0.0
442	295	Good	0.28	51.2	184.0	0.69	127.0	23.6	0.0

Table 7-1. (Continued)

Plot	Wetland	Condition	Dry Density (g/cm^3)	P g/m^3	P g/Mg	Sediment g/trap	P loading rate g/Mg	% OM	% CCE
442	301	Good	0.41	70.1	169.8	0.79	134.1	13.0	0.0
498	146	Good	0.69	39.7	57.6	8.92	513.7	6.7	0.0
498	277	Good	0.60	41.2	68.2	3.22	219.6	7.5	0.0
Mean	Good					3.4	259.0	15.1	
Mean	Poor					3.0	291.7	9.4	
SD	Good					3.4	217.6	7.4	
SD	Poor					2.0	209.2	2.4	

Table 7-2. **Estimation of soil loss in four CWLSA wetlands plus a non-eroded control site using Cs-137 analysis (Soileau et al. 1990). Bulk density values are mean values from three field samples, Cs-137 activities are mean values of two laboratory runs. Each Cs-137 sample was a composite of three basin subsamples. Gamma ray count time was 57600 seconds.**

	P1	T1	P7	C7	Non-eroded"
Bulk Density (Mg/m^3)	0.83	1.17	0.89	1.39	1.20
Cs-137 Activity Bq/m^2	5.01	2.99	2.20	1.11	3.33
Soil Loss (-) or gain (+) (metric tons	+16.13	-4.56	-11.61	-35.64	0.00

Combining P with one or both of the other two variables does not improve the P-value, or seemingly add to the separation of the two condition groups. Distances between phosphorus values in the poor-condition group are about twice the distances in the good group, indicating unequal variances in the two groups of data, one of the reasons for using MRPP.

7.4.4 Results of 1992 and 1993 EMAP Soil Characterization

Poor wetland condition included land recently placed into CRP, since the soil analyses probably reflect long-term conditions in the wetland basin. Least squares (LS) means and mean comparisons were made from log-transformed data Tables 7-5, 7-6, and 7-7. Log-transformation was necessary on all but the pH variables, due to data distributions being skewed to the left. Back-transformed means are also shown in those tables.

Nitrate varied significantly with year ($F_{1,95}=9.46$; $P=0.0027$), and with depth ($F_{1,48}=13.559$; $P = 0.0006$). The year effect implies that the differences between 1992 and 1993 are consistent for condition, zones and depths (i.e., comparisons between years can be made by ignoring condition, zone and depth). The depth effect implies that depth differences are consistent between condition, zones, and years. Phosphorus varied significantly with year ($F_{1,95}=5.02$; $P=0.0274$), zone ($F_{2,35}=6.57$; $P=0.0038$) and marginally with condition and depth interaction ($F_{1,48}=2.88$; $P=0.0962$). OM only varied significantly with depth ($F_{1,48}=108.69$; $P=0.0001$).

Table 7-3. **CWLSA Soil nutrient analysis of 0-15 cm samples from wetlands P1, T1, P7, and C7, Cottonwood Lake Study Area, 1992. Abrev.: WM = wet meadow, SM = shallow marsh, NO_3^- = nitrate. Units: OM = % mass, NO_3 and P g/m3.**

Wetland	Transect	Condition	Zone	OM	NO_3	P
P1	1	Good	WM	4.3	8.1	6.9
P1	2	Good	WM	4.2	9.9	7.4
P1	3	Good	WM	2.4	6.2	5.0
P1	4	Good	WM	2.6	6.3	3.8
P1	5	Good	WM	1.9	5.0	4.4
P1	6	Good	WM	2.2	5.0	5.6
P1	7	Good	WM	5.1	13.7	7.5
P1	8	Good	WM	3.2	6.3	5.0
P1	9	Good	WM	2.6	3.8	6.3
P1	10	Good	WM	3.1	5.6	6.9
T1	1	Good	WM	4.9	6.3	8.1
T1	2	Good	WM	6.0	5.0	6.2
T1	3	Good	SM	8.9	13.7	34.3
T1	4	Good	WM	6.4	11.2	6.9
P7	1	Poor	SM	9.5	13.8	33.8
P7	2	Poor	SM	9.8	17.4	26.1
P7	3	Good	WM	11.7	11.9	11.3
P7	4	Good	WM	6.3	8.1	7.5
P7	5	Good	WM	6.9	11.2	7.5
P7	6	Poor	WM	7.9	8.2	8.8
C7	1	Poor	WM	9.9	13.7	35.0
C7	2	Poor	WM	2.2	1.9	11.3
C7	3	Poor	WM	9.4	6.3	20.7
C7	4	Poor	WM	9.6	11.3	35.2

Table 7-4. **MRPP statistical analysis of soil nutrient data, CWLSA, 0-15 cm soil depth, 1992.**

Variable	Condition	Distance	Observed Delta	P-Value
Phosphorus	Good	4.96		
	Poor	13.3	7.39	0.00047
Phosphorus and Nitrate	Good	7.07		
	Poor	15.30	9.47	0.00071
Nitrate	Good	3.70		
	Poor	6.38	4.48	0.26474
Phosphorus Nitrate and Organic Matter	Good	8.01		
	Poor	15.74	10.27	0.00071
Organic Matter	Good	2.93		
	Poor	2.58	2.82	0.00319
Phosphorus and Organic Matter	Good	6.38		
	Poor	13.88	8.57	0.00049

EC varied significantly with zone ($F_{2,35}$=3.42; P=0.044) and with depth ($F_{1,48}$=3.93; P=0.0531). pH varied with year ($F_{1,95}$=3.66; P = 0.0588), zone ($F_{2,35}$=3.93; P=0.0289), and depth ($F_{1,48}$=13.59; P = 0.0006). Sand ($F_{2,33}$=21.19; P=0.0001), silt ($F_{2,33}$=5.42; P=0.0092) and clay ($F_{2,33}$=7.79; P=0.0017) varied significantly only with zone. All other effects and interactions were nonsignificant.

Tables 7-5 and 7-6 show Fisher's protected LSD tests and LS means for log-transformed NO_3 and P data. The letters a, b, and c located next to the LS Mean values indicate whether or not the means are significantly different. Table 7-7 shows Fisher's protected LSD tests for sand, silt, and clay. Values followed by another value with a common letter, for example shallow marsh and wet meadow silt in Table 7-7, are not significantly different. Significant differences by year and depth increment occurred for nitrate values. The 7.5 and 30 cm depths shown in the tables indicate the midpoint of the 0-15 cm and 15-45 cm depth increments we sampled in the field. Significant differences of P levels exist by year, by zone, and by condition*depth interaction. Higher P concentrations occurred in 1993, and concentrations were highest in the deep marsh zone. The condition*depth interaction indicates the P concentrations in the 0-15 cm depth soils in the poor-condition wetlands were significantly higher than in the good-condition soils of the same depth increment, independent of zone or year.

OM mean concentrations (Table 7-6) were significantly higher in surface soils compared to subsoils, independent of other interactions. This is usually the case in soils, where decaying surface vegetation and microbial activity leave higher concentrations of OM near the surface. EC (Table 7-6) was significantly higher in deep marsh zones (DM) than in either shallow marsh (SM) or wet meadow (WM) zones. Wet meadow and shallow marsh salinity (EC) were not significantly different. The Fisher's test also found significant differences in pH values by year, by zone, and by depth (Table 7-6). Clay content was highest in deep marsh zones (Table 7-7).

7.4.5 CWLSA Soil Oxidation-Reduction Potential

All but one (0.00 being neutral) of the soil oxidation-reduction measurements from September and October 1992 are positive, indicating relative oxidizing conditions (Table 7-8). May and June 1993 values are mixed, but mostly negative, indicating relative reducing conditions. Interior zones of T7 and P7 flooded deep enough to prevent us from monitoring soil oxidation-reduction. Wetland P1 was not monitored.

7.5 EVALUATION

7.5.1 Seasonal Sedimentation

Results of the chemical analyses of the trapped sediments did not reveal significant differences in P inputs occurring between the sampled good- and poor-condition wetlands. Organic matter makes up a larger proportion of the sediments in good-condition wetlands than in the poor-condition wetlands (8.3 g/100 g). A greater proportion of the poor-condition sediment is mineral material. This is a potentially important indicator reflecting higher rates of erosion and sedimentation in the poor-condition wetlands. While organic matter will, for the most part, be decomposed to biologically recyclable nutrients and gases, the inorganic sediment will remain mostly inert and, given time, fill in the wetland.

7.5.2 Long-term sedimentation

The Cs-137 study addressed the problem of sedimentation indirectly by examining soil loss from the wetland watershed side slopes. The C7 wetland had the highest rate of soil loss. However the value for P1 indicates Cs-137 and soil deposition on its side slopes.

Table 7-5. Least squares means for nitrate and phosphorus, EMAP sample wetlands, 1992-93.

Variable	Effect		LS Mean[a] ln(Y+1)		Back-transformed LS Mean	
Nitrate	YEAR	1992	1.88 b		7.55	
		1993	1.42 a		5.14	
		Pooled MSE =	0.2508			
	DEPTH (cm)	7.5	1.80 b		7.05	
		30	1.49 a		5.44	
		Pooled MSE =	0.1563			
Phosphorus	YEAR	1992	3.26 a		27.05	
		1993	3.44 b		32.19	
		Pooled MSE =	0.1348			
	ZONE	DM	3.62 (40) c		38.34	
		SM	3.34 (86)		29.22	
		WM	3.08 (159) a		22.76	
		Pooled MSE =	0.4759			
			DEPTH (cm)		DEPTH (cm)	
	CONDITION		7.5	30	7.5	30
		High	3.36 (78)a	3.09 (78)a	29.79	2.98
		Low	3.75 (65)b	3.19 (64)b	43.52	25.29
		Pooled MSE =	0.0926			

[a] Within a column, LS Means followed by a common letter are not significantly different at the 0.05 level using Fisher's protected LSD value.

Table 7-6. **Least squares means for log transformed percent organic matter (OM) and electrical conductivity (EC), EMAP sample wetlands, 1992-93. pH data were not log transformed.**

Variable	Effect		LS Mean[a] 1n(Y+1)	Back-transformed LS Means
% OM	DEPTH (cm)	7.5	2.10 (143) b	9.17
		30	1.64 (142) a	6.16
		Pooled MSE =	0.0662	
EC (micromhos)	ZONE	DM	6.75 (40) b	855.06
		SM	6.33 (86) a	562.16
		WM	6.19 (159) a	488.85
		Pooled MSE =	0.6464	
	DEPTH (cm)	7.5	6.38 (143)	590.93
		30	6.46 (142)	640.06
		Pooled MSE =	0.0681	
pH	ZONE	DM	7.15 (40) b	----
		SM	6.94 (86) a	----
		WM	7.12 (159) b	----
		Pooled MSE =	0.1476	
	YEAR	1992	6.96 (143) a	
		1993	7.18 (142) b	
		Pooled MSE =	0.1924	
	DEPTH (cm)	7.5	6.96 (143) a	
		30	7.18 (142) b	
		Pooled MSE =	0.1228	

[a] Within a column, LS Means followed by a common letter are not significantly different at the 0.05 level using Fisher's protected LSD value.

Table 7-7. **Least squares means for log-transferred percent sand, silt, and clay; EMAP sample wetlands, 1993.**

Variable	Effect		LS Means 1n(Y+1)[a]	Back Transformed LS Mean
Sand	ZONE	DM	2.60 (11) a	14.46
		SM	3.22 (22) b	26.02
		WM	3.45 (39) c	32.5
Silt	ZONE	DM	4.20 (11) b	67.69
		SM	3.92 (22) a	51.40
		WM	3.85 (39) a	47.99
Clay	ZONE	DM	3.32 (11) b	28.66
		SM	2.95 (22) a	20.11
		WM	2.81 (39) a	17.61

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[a] Within a column, LS Means followed by a common letter are not significantly different at the 0.05 level using Fisher's protected LSD value.

Table 7-8. **Soil oxidation-reduction potential measurements from 3 CWLSA wetlands. September, 1992-June, 1993.**

Date	Wetland	Zone	Rep	Depth (cm)	(mVolts)
09-10-92	T1	WM	1	45	+235
			1	15	+196
		SM	1	45	+210
			1	15	+240
10-07-92	T1	WM	1	45	+311
			1	15	+301
		SM	1	45	+318
			1	15	0.00
05-13-93	T1	WM	1	45	+148
			1	15	+200
		SM	1	45	-428
			1	15	-335
06-17-93	T1	WM	1	45	-107
			1	15	-170
		SM	1	45	-69
			1	15	-46
09-10-92	T7	WM	1	45	+350
			2	45	+374
			1	15	+365
			2	15	+337
	T7	SM	1	45	+282
			2	45	+340
			1	15	+310
			2	15	+313
10-07-92	T7	WM	1	45	+335
			2	45	+380
			1	15	+442

Table 7-8. (Continued)

Date	Wetland	Zone	Rep	Depth (cm)	(mVolts)
			2	15	+418
	T7	SM	1	45	+345
			2	45	+393
			1	15	+410
			2	15	+420
05-13-93	T7	WM	1	45	+10
			2	45	-265
			1	15	-254
			2	15	-200
		SM	FLOODED		
06-17-93	T7	WM	1	45	-159
			2	45	-250
			1	15	-152
			2	15	-208
		SM	FLOODED		
09-10-92	P7	WM	1	45	+270
			2	45	+280
			1	15	+207
			2	15	+154
	P7	SM	1	45	+253
			2	45	+246
			1	15	+248
			2	15	+245
	P7	DM	1	45	+250
			1	15	+245
10-07-92	P7	WM	1	45	+290
			2	45	+272
			1	15	+254
			2	15	+264
	P7	SM	1	45	+278
			2	45	+288
			1	15	+314
			2	15	+317
			2	45	+201
05-13-93	P7	WM	1	45	+195
			1	15	+217
			2	15	+180
	P7	SM	1	45	-144
			2	45	+107
			1	15	-302
			2	15	-272
	P7	DM	FLOODED		
06-17-93	P7	WM	1	45	-205
			2	45	-120
			1	15	+214
			2	15	+260
	P7	SM	1	45	-504
			2	45	-510
			1	15	-460
			2	15	-480

The higher Cs-137 values on P1 slopes could be due to snowcatch on the grass-covered side slopes, with the snow containing Cs-137 fallout. Snow particles nucleate around dust particles in the air and thus would be added to soil upon melting. Another possibility is that explaining the apparent soil deposition on P1 side slopes of P1 is soil loss from the flat, "noneroded" site. We see no evidence, however, to support this. Another possibility, slow soil creep on the P1 side slopes may have moved Cs-137-attached soil from higher on the hill slope down to and on top of the mid-slope position where we collected our samples. Finally, the problem of apparent soil deposition of the P1 side slope could be explained by inadequate sample size.

7.5.3 Soil Characterization: CWLSA and EMAP Studies

Soil NO_3 was significantly lower in 1993 than in 1992. At least three factors might explain this contrast: denitrification, leaching, and change in sample handling procedures. Summer 1992 marked the end of a drought in the northern plains. Relatively high precipitation during the second half of 1992 and first half of 1993 refilled previously dried wetlands. Denitrification occurs as a result of chemical reduction usually associated with saturated, anaerobic environments. Nitrate, an oxidized nitrogen (N) compound is converted to more reduced compounds including ammonia (NH_3), nitrous oxide (N_2O), and nitrogen (N_2), all three of which are gaseous and return to the atmosphere. This natural process may have produced lower NO_3 levels in 1993. NO_3 is also highly soluble and may have leached to deeper soil depths in 1993. Finally, the lower NO_3 values in 1993 may have been due to a procedural change. In 1993, we dried soil samples overnight in an evaporating oven. This relatively fast drying was done to help eliminate possible oxidation of formerly reduced N compounds upon exposure to air. In 1992, samples had longer exposure to air that could have caused NO_3 values to be elevated.

NO_3 varied significantly with depth, the higher concentrations being in the 0-15 cm samples. This is most likely due to the greatest portion of the total soil nitrogen pool's association with soil organic matter. Decomposition and oxidation of organic nitrogen produces higher NO_3 concentrations in the topsoil.

From the studies at the CWLSA and EMAP sample wetlands, P was the strongest indicator showing the apparent impact of cultivation on wetland nutrient concentrations. Phosphorus, unlike nitrogen, has no stable gaseous forms and, once in the wetland, tends to remain there. Although nitrogen fertilizer is commonly applied to cultivated fields, denitrification under reducing conditions results in reduced gaseous forms of nitrogen escaping to the atmosphere.

Considerable evidence exists linking P loading to runoff and soil erosion. Andraski et al. (1985) compared P losses in runoff from four different tillage schemes including conventional till, chisel plow, till-plant, and no-till. They measured total P, dissolved molybdate-reactive P (DMRP) and algal-available P (AAP). It is important to note here that there are differences in soil P forms. Some are more available to plants than others. The DMRP and AAP are of higher ecologic importance than total phosphorus, since much of the total P is tied up in relatively insoluble minerals. Conservation tillage greatly reduced total P and sediment runoff. DMRP runoff from conservation tilled plots was lower than or equal to runoff from the conventionally, tilled plots. The AAP runoff comparisons, perhaps the best indicator of P pollution, were highest for the conventionally tilled plot and the till-plant plot, and lowest in the no-till plot. Andraski et al. (1985) stated that the algal-available P includes the dissolved (DMRP) phosphorus and about 20% of the total P. AAP is made up of about 30% DMRP, and 70% is inorganic, easily desorbed or easily dissolved particulate P associated with the sediment. The important point here is that the algal available P is mostly particulate phosphorus.

Since soil P can take many forms which might play important ecological roles, we need to know which ones we are measuring when performing a soil test. Wolf et al. (1985) examined different soil tests, including the Bray-I, Olsen, and Mehlich I methods to see how well they corresponded to equilibrium-dissolved-P concentration, "labile" P, and AAP. For north-central soils, the Olsen $NaHCO_3$ soil test P related significantly to the AAP.

Organic matter varied significantly with depth, as was expected. Under natural soil conditions, organic matter decreases with depth. Under extreme cases of erosion and sedimentation, however, where mineral soil is deposited on top of more organic-rich topsoil, organic matter increases with depth. Such a profile may be a good indicator of disturbance.

Although pH and EC do not appear to be valid indicators of wetland condition, they do provide an interesting reflection of climatic changes in the region. In our study pH was shown to be significantly different between years, but this fact, we are quite sure, is due to a laboratory procedural change in the way we measured pH. In 1992, we measured pH in 0.01M $CaCl_2$, which forces H^+ ions off soil exchange sites and, subsequently, lowers the measured pH value. In 1993, we measured pH in distilled water. By doing so, we could use the same sample prepared for the EC analysis to measure pH. This saved time and reduced the total amount of soil needed to carry out the full set of lab analyses. Our data tell us the soil pH measured in $CaCl_2$ is 0.1 to 0.4 units lower than the same soil measured in distilled water.

The soil texture analyses showed significant differences between zones, as expected. Coarser soil textures are found along the wetland edges, finer textures in the wetland interiors. Since less energy is required to transport silt or clay, wave action along wetland edges sorts fine particles and relocates them to deep water, leaving the edges sandy; i.e., creates a beach.

Soil classification, which is not easily quantified, appeared to reveal more information about the climatic and hydrologic conditions in wetlands than about land use. A notable exception, however, occurred at wetland C7 at the CWLSA. In the wet meadow of C7, we found an apparent buried A horizon and "cumulic," dark-colored A horizons in all 4 wet meadow profiles. Cumulic A horizons are relatively unusual in wet meadow sites where wave action generally sorts fine-grained soil material and organic colloids. The other wet meadow profiles we classified in other wetland basins at the CWLSA had much thinner and sandier A horizons than those at C7. We interpret this difference to relatively rapid soil deposition on the C7 wet meadow.

For the soils sampled from the wetlands in the 10.4 km^2 plots, generally, the eastern wetlands were non-calcareous Endoaquolls and Argiaquolls, and the more western wetlands were, more calcareous Typic Calciaquolls, Cumulic (Calc) Endoaquolls and Aeric Calciaquolls. In some cases, apparent changes in classification from 1992 to 1993 within the same zone (e.g., wet meadow in wetland 442-295, Appendix 7-1), reflected the drought-to-deluge transition that occurred over much of the northern prairie from 1992 to 1993. Classification of Mollisols often hinges on the presence or absence of leachable constituents, especially ($CaCO_3$). In upland soils, where runoff occurs and leaching rate is relatively low, calcareous soils are probably stable through typical northern prairie climatic fluctuations. However, where water is focused in the landscape, i.e., in the wetlands, leaching is apparently capable of removing enough $CaCO_3$ from a soil profile to alter its classification.

7.5.4 Soil Oxidation-Reduction

Soil oxidation-reduction is a function of temperature, microbial activity, and the availability of elements or compounds that can serve as electron acceptors during metabolic activities. Under aerobic conditions, oxygen is the primary electron acceptor, but when oxygen is depleted, other compounds including nitrate, manganese oxides, iron oxides, sulfate, and carbon serve the microbial community as electron sinks. Oxidation and reduction of iron oxides in water-logged soils produces observable "redoximorphic" patterns (commonly known as mottles) we use to assess the hydrologic characteristics of soil subject to saturation. In 1992 we had experienced 6 years of drought and had questions about whether the redoximorphic features we were seeing at CWLSA were due to contemporary oxidation-reduction processes, or whether the morphology was relict. The monitored sites became mostly saturated and reduced in the spring of 1993. The sites were under several feet of water by late summer, 1993 and undoubtedly anaerobic. At that point we know that during reduction iron is mobilized.

Although important in understanding the link between soil morphology and biogeochemical processes, the oxidation-reduction data we collected do not appear to be valid condition indicators.

7.6 FUTURE RECOMMENDATIONS

Soil P is the best indicator of wetland condition based on results from our study, specifically, P found in the 0-15 cm depth. We recommend the continued use of $NaHCO_3$-extractable P (Olsen et al. 1954, Knudsen and Beegle 1988) when testing for biologically available P in northern prairie wetlands. This test, also commonly known as "Olsen phosphorus" is routinely performed at relatively low cost at the NDSU Soil Fertility Laboratory.

Organic matter and texture analyses, although not statistically significant variables in this study, remain potential indicators of severe soil disturbance, and we recommend they be included in future studies.

EC can show a relationship between fluctuating climate and landscape salinity, but based on our data we do not recommend it as an indicator of wetland condition. Salinity is a water quality issue in parts of the PPR, and it may be useful to track long-term precipitation-soil salinity patterns to better understand how landscape salinity responds to climatic variations. An inexpensive 1:1 soil-water suspension EC test might be useful for this purpose.

In addition to our doubts about using Cs-137 because of the confounding effect of cultivation, it is too expensive. Our data indicate a relatively high number of samples from each basin would be needed to develop accurate results for deposition rates in PPR wetlands. Each gamma sensor can count only 1 or 2 samples a day, depending on its sensitivity and the amount of Cs-137 in the soil. Further, since fallout from the 1963-64 maximum is already half of its original activity, we will have less Cs-137 activity to work with as time goes by. Therefore, we do not recommend Cs-137 for long term monitoring in the PPR.

We recommend dropping the following variables from indicator testing: NO_3^-, pH, calcium carbonate equivalent test, soil classification, soil oxidation-reduction and Cs-137.