Ecological changes in two contrasting lakes associated with human activity and dust transport in western Wyoming

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Abstract

The atmospheric transport and deposition of aerosols has the potential to influence the chemistry and biology of oligotrophic alpine lakes. In recent decades, dust and nitrogen emissions to alpine ecosystems have increased across large areas of the western USA, including Wyoming. Here, we use sediment geochemistry and $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotopes to examine historical dust deposition rates to alpine lakes in the southwestern region of the Wind River Range, Wyoming. We evaluate the biological response using diatom fossil assemblages and sediment pigment concentrations. Sediment core analyses indicated that prior to a recent rise in dust flux, phosphorus concentrations and species composition were similar to those found in other alpine lakes in the region. Concomitant with a ~50 fold increase in dust flux to the sediments circa 1940, sediment proxies revealed a 2-3 fold increase in normalized sediment phosphorus content, an increase in the diatom-inferred total dissolved phosphorus concentration from ~4 to 9-12 $\mu g\ L^{-1}$, a tenfold increase in diatom production, and a relative increase in cyanobacteria abundance. The increase in dust influx during the 20th century appears to be due in part to human factors and demonstrates the potential for dust and other atmospheric pollutants to significantly alter remote aquatic ecosystems.

Introduction

Human land-use including agriculture, grazing, and industry are altering regional biogeochemical cycles through elevated emissions of nitrogen compounds (Galloway et al. 2003) and dust to the atmosphere (Neff et al. 2008). Mountain ranges act as natural barriers to atmospherically transported material where deposition may occur as dry fall or dissolved in precipitation. Because alpine lakes are characteristically nutrient-poor and derive the bulk of their nutrients from atmospheric sources, these lakes are very sensitive to variation in atmospheric deposition (Psenner 1999; Pulido-Villena et al. 2006; Mladenov et al. 2009).

Over the last two decades, dust emissions have increased in many regions of the western US, in some areas by up to 400% (Brahney et al. 2013). The effects of dust deposition on alpine lakes can be challenging to determine because dust is not routinely measured, and because nitrogen (N) and dust deposition may be co-occurring in a variety of western US locations. Many mountain ranges in western states are downwind of urban, industrial, and agricultural sources of N as well as arid and semi-arid regions that regularly produce dust storms (Fenn et al. 2003; Painter 2007). The influence of nitrogen deposition on alpine lake ecology has been investigated in several regions of the US and has been shown to cause large changes in ecological and biogeochemical processes (Baron 2000; Elser et al. 2009). Relatively less is known about the contribution of nutrients and alkalinity from dust including the deposition of phosphorus (P) and calcium carbonates to aquatic ecosystems (Morales-Baquero et al. 2006; Pulido-Villena et al. 2006).

In the mountain ranges of the interior western US and in other mountain ranges near arid and semi-arid regions, the erosion of desert and agricultural soils has the potential to bring nutrients, particularly phosphorus (within soils, organisms, and phosphate minerals), acid-neutralizing minerals (carbonates), and heavy metals to alpine lakes. Measurements of dust chemistry have shown that dust can transport appreciable amounts of phosphorus and carbonates to depositional regions (Lawrence and Neff 2009). Though there are relatively few studies on the effects of dust deposition to lakes, dust-associated

increases in phosphorus deposition have been linked to altered lake ecosystems. In the Sierra Nevada Mountains in Spain, dust phosphorus was found to increase chlorophyll-*a* concentrations, bacterial abundance, and modify plankton diversity (Pulido-Villena et al. 2008; Reche 2009). A recent study in the Wind River Range found high dissolved phosphorus concentrations associated with elevated dust deposition to lakes proximal to the local dust source (Brahney et al. 2014a). These dust-affected lakes had a unique species composition and phytoplankton and zooplankton biomasses were two orders of magnitude greater than found in lakes elsewhere in the area. Total dissolved phosphorus (TDP) measurements were as high as 14 $\mu g\ L^{-1}$ in the dust-affected lakes, an unusually high concentration for alpine lakes in rocky granite catchments.

Though recent regional-scale analyses indicate a rise in dust emissions across large regions of western USA (Brahney et al. 2013), it is not clear how large these increases are relative to historical rates in many arid and semi-arid regions. Similarly, there is little information regarding the biological importance of these changes. Here we select two lakes for multi-proxy sediment core analyses from two locations in the Wind River Range to 1) document the historical contribution of dust to alpine lake ecosystems in the southwestern (dust-affected) portion of the range; 2) determine if the unique plankton species community in southwestern Wind River lakes are a result of naturally higher atmospheric nutrient contributions; and 3) evaluate the effects of dust deposition while evaluating the coincident effects of nitrogen deposition based on measured deposition records from four sites around the range.

Site descriptions

The Wind River Mountains are located in west central Wyoming (Figure 1). The dominant dust source is from the Green River Basin located to the south and west of the range. Due to the proximity to the dust source, deposition rates are greatest in the southwestern region, and have been measured at an average rate of 4 $g\ m^{-2}\ yr^{-1}$ and a maximum rate of 11 $g\ m^{-2}\ yr^{-1}$. Dust deposition rates diminish towards the north, where measured rates are around 1 $g\ m^{-2}\ yr^{-1}$ (Dahms 1993; Brahney et al. 2014a).

Data from the National Atmospheric Deposition Program (NADP) indicate that N deposition increased similarly at all measured locations since record keeping began in the mid 1980's. NADP site locations (WY06, WY02, WY97, WY98) are shown in figure 1. No major urban centers exist in western Wyoming; however, the area is extensively used for rangeland, oil & gas, and mining operations. Settlement in the area expanded after the Homestead Act of 1862 and the discovery of precious metals in the region (Hausel 1994; Cassity 2011). The Green River Basin also contains one of the world's largest Trona/Soda Ash deposits (Dyni 1996), the first mine shaft was drilled in 1946 and the industry expanded in the 1970's (Blm 2014). The Pinedale Anticline, located in the Green River Valley adjacent to the Wind River Range, is the 3rd largest gas field in the US. The first well was drilled in the 1920's but the industry did not begin to grow appreciably until the 1950's, expanding considerably in the 1980's and again in 2000. From 2000-2008, 2683 wells were emplaced and NO_x emissions increased from 3000 to 7800 cubic tons, peaking in 2006 (Wo&Gcc 2011). These operations, and others in western Wyoming, are a source of nitrogen to the Wind River Mountains (Mcmurray et al. 2013).

The Wind River Mountains consist predominately of Archean granitic rocks, granitic gneisses and migmatites (Frost et al. 1998) that are distinctly different from the Eocene sedimentary sequences found in the Green River Valley, which are composed of sandstone, siltstones, limestones, evaporates, and volcanic ash layers (Sullivan 1980). The geologic differences make isotopic, mineralogical, and geochemical separation of the Green Valley dust source from the local bedrock possible. Dahms and Rawlins (1996) used the presence of volcanic minerals in soils not found in the prevailing geology of the Wind River Mountains to document the presence of Green Valley dust across the range.

Dust from the Green River Valley has a high concentration of phosphorus at 3.0 mg g^{-1} (Brahney et al. 2014a), relative to the global average dust concentration of 1.1 mg g^{-1} (Lawrence and Neff 2009). These high values are due to both the sizeable concentration of organic material in the dust (40%) and to the presence of the mineral apatite ($Ca_5(PO_4)_3OH$) (Dahms 1993). While phosphorus from the organic fraction is readily available to contribute to the nutrient pool, apatite phosphorus is not generally

considered a significant source of phosphorus to lakes (Wetzel 2001). However, it has been shown that algae and bacteria can solubilize natural apatite at higher pH levels, especially if the apatite is made available at smaller grain sizes (Smith et al. 1977). Therefore, the fine-grained apatite from atmospheric deposition is a potential source of P to aquatic communities.

Of the lakes analyzed in Brahney et al. (2014a), we selected two lakes for this analysis that have markedly different dissolved phosphorus concentrations. North Lake (3085 meters above sea level) a dust-affected lake on the southwestern side of the divide has an average TDP concentration of 12 $\mu g\ L^{-1}$, while Lonesome Lake (3093 meters above sea level) on the southeastern side of the continental divide has a lower average TDP concentration at 4 $\mu g\ L^{-1}$ (Brahney et al. 2014a). Particulate P as well as dissolved and particulate calcium (Ca) concentrations are greater in North Lake than in Lonesome Lake (Table 1). Both regions have undergone recent small increases in N deposition from less than 0.5 kg $ha^{-1}\ yr^{-1}$ to 1.5 kg $ha^{-1}\ yr^{-1}$ (Nadp 2009). North Lake has no geographic barrier to the major dust sources upwind and is the site most geographically predisposed to dust deposition, whereas Lonesome Lake is located on the eastern side of the divide and presumably more protected from westerly winds by the large granite towers that delineate the continental divide in the Wind River Range. Both lakes catchments are dominantly composed of bare rock. North Lake's catchment is 92% bare rock, with the remaining area composed to shrub vegetation, whereas Lonesome Lake is surrounded by a small wooded area, though 80% of the catchment remains rocky and un-vegetated.

Methods

Coring

To obtain long-term records, lake sediment cores from North and Lonesome lakes were collected in the summer of 2008 using a percussion coring system. Cores were taken from the deepest area of each lake as determined by a Sound Navigation and Ranging (SONAR) instrument. Two cores were retrieved

from Lonesome Lake (LSM08-1, LSM08-2) at 15 m depth, which were 50 and 144 cm long respectively. In North Lake two cores were retrieved (NOR08-1, NOR08-2) from 9 m depth, which were 48 and 100 cm in length. The percussion cores were shipped to the National Lacustrine Core Facility at the University of Minnesota where they were split, photographed, and analyzed for bulk physical properties including density, loss on ignition (LOI), and magnetic susceptibility. Due to disturbance of the surface sediments that typically occur through the percussion coring method, we also obtained sediments from the same lakes in the summer of 2009 using a gravity coring system, NOR09-2, which was 11 cm long, and LSM09-2, which was 14.5 cm long. Gravity cores were sectioned on site at 0.25 cm intervals from 0-3 cm, and at 0.5 cm intervals for the remainder of the cores. Sediments were bagged in labeled Whirl-Pak bags and transported on ice to the University of Colorado.

Dating

Gravity cores (LSM09-2, NOR09-2) were dated using ^{210}Pb at MyCore Laboratories in Ontario Canada. Core interval ages were calculated assuming a constant rate of supply (CRS) of unsupported ^{210}Pb (Appleby and Oldfield 1978). Percussion cores (LSM08-2, NOR08-2) were dated using radiocarbon and $^{239+240}$Pu isotopes (Ketterer et al. 2004). Radiocarbon samples were carefully extracted using acid-washed foreceps, cleaned with DI water, and dried before shipment to the USGS National Center where AMS ^{14}C analysis was performed. $^{239+240}$Pu isotope analysis was conducted at the University of Northern Arizona. We used the CLAM software for classical non-Bayesian locally weighted spline age-depth modeling (Blaauw 2010). CLAM uses IntCal09 and depth-ages are weighted to their calibrated age probabilities. NOR08-1 and NOR08-2 were compared using stratigraphic markers including density peaks that corresponded to visible lighter colored laminae (Figure 2). Density peak depths were correlated between cores (r^2=0.99), allowing the incorporation of radiocarbon ages from NOR08-1 into the NOR08-2 model. We ascribed the coring date to the surface sediment values for NOR08-2. Although considerable mixing occurred in the upper sediments, there was headspace in the coring apparatus at the time of coring

and we therefore do not believe that sediment was lost. We cannot state the same for LSM08-2; however, based on the density, LOI, and MS profiles between LSM08-1 and LSM08-2, and because the coring rope was measured to the lake depth, the loss does not appear to be more than a few centimeters. Regardless, for this reason it was not possible to ascribe the coring date to the surface sediment of LSM08-2. This leads to relatively greater uncertainty in surface ages for the LSM08-2 age model. However, all gravity cores were obtained with minimal surface disturbance as evidenced by the intact sediment surface-water interfaces. The corresponding sediment mass accumulation rates for each core (g cm^{-2} yr^{-1}) are then derived from the density (g cm^{-3}) data. Where available, we derived the mass accumulation rates for the mineral and organic fractions by multiplying the sedimentation rate at each interval by the fractional mass of either component based on the LOI data.

Dust deposition history

We use neodymium ($^{143}Nd/^{144}Nd$) and strontium ($^{147}Sm/^{144}Nd$, and $^{87}Sr/^{86}Sr$) isotope ratios from dust and local bedrock samples to separate exogenous from local material in the North and Lonesome Lake sediments (Grousset and Biscaye 2005; Lawrence 2010). For comparison to other studies, present-day $^{143}Nd/^{144}Nd$ ratios have been normalized as follows, $\varepsilon_{Nd} = (^{143}Nd/^{144}Nd / 0.512638)-1)*10^4$. The potential for isotopic separation arises because the catchment bedrock is Archean in age (Koesterer et al. 1987) while the dust region in the Green River Valley is composed of much younger Eocene volcanic deposits and lake beds (Sullivan 1980). Strontium-87 is produced from the beta decay of ^{87}Rb; over time, the amount of ^{87}Sr increases and older rocks generally have higher $^{87}Sr/^{86}Sr$ ratios (Capo et al. 1998). Similarly, ^{143}Nd is produced from the alpha decay of ^{147}Sm and older rocks tend to have lower $^{143}Nd/^{144}Nd$ ratios. The continental crust is generally light in rare earth elements relative to the upper mantle, so ε_{Nd} values for the continental crust are negative and decrease with increasing crustal age.

Because ^{147}Sm and ^{87}Rb have relatively long half-lives, the measured isotope ratios used here can be considered as conservative tracers.

Dry deposition was sampled using bulk deposition samplers that were placed in triplicate at the nearby USFS atmospheric monitoring station at Black Joe Lake (3130 meters above sea level), in the southwestern region of the range (Figure 1). Samplers were composed of polyethylene funnels that were 16 cm in diameter and filled with glass marbles to effectively capture dry deposition. The funnels were mounted on posts 2 m above ground. The base of the funnel was fitted with Tygon tubing allowing the sample to drain into opaque brown Nalgene bottles. The funnels were fitted with a bird ring twice the diameter of the funnel to discourage birds from landing on the sampler itself. A small piece of nylon mesh was fitted in the drain to prevent contaminants from entering the bottle. The sample bottles were exchanged every 1-4 weeks after rinsing the apparatus with 50 mL of DI water and kept frozen until analysis. The samples were pooled and evaporated to retain dust for isotope analysis. We removed salts that precipitated during the drying process via sediment washing, aspirating, and re-drying. Dust and lake sediments were initially ashed and leached with HCl to remove organic matter and other labile mineral fractions. Digestion of all samples, bedrock, dust, and sediment for isotope analysis follows the methods outlined in Pin et al. (1994). Neodymium and strontium isotope analyses were completed at the Thermal Ionization Mass Spectrometry Lab at the University of Colorado in Boulder using a Finnigan-MAT 6-collector solid source mass spectrometer following procedures outlined in Farmer et al. (1991). Analytical precision for 9 measurements of the SRM-987 $^{87}Sr/^{86}Sr$ standard during the analyses yielded a 2σ range of $\pm 2x10^{-5}$ ‰, and for 15 measurements of the La Jolla $^{143}Nd/^{144}Nd$ standard a 2σ range of $\pm 8x10^{-6}$ ‰. The concentration of dust in sediment intervals is estimated in two ways, by 1) using the average of mass-fraction calculations from $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotope data and 2) using a constrained least-squares equation using the isotope data with the dust and bedrock fractions as end-members. Constraints are such that the sum of the bedrock and dust proportions must equal one. Dust deposition fluxes through the cores are estimated by multiplying the dust fraction by the mineral sedimentation rate at each interval. Note

that, because dust in the lake sediments includes that from the catchment that was focused into the lake, this represents a catchment-integrated flux.

Sediment Geochemistry

Bulk geochemical analysis was performed on North Lake cores (NOR09-2, NOR08-2) and the Lonesome Lake percussion core (LSM08-2) and surface sediment geochemistry is available for the gravity cores (NOR09-2) and (LSM09-2) (Brahney et al. 2014a). Sediment samples were digested using trace element grade HCl, HNO_3 and HF, and H_2O_2 (EPA, 1996) and analyzed for major and minor element concentrations at the Laboratory for Environmental and Geological Studies (LEGS) at the University of Colorado. Samples were analyzed using a Perkin Elmer Elan DRC-E ICP-MS and an ARL 3410+ ICP-AES, analytical precision is within 5% for both instruments. We evaluated changes in sediment composition by using elemental ratios. Redox cycling within the sediments tends to produce an upcore increase in redox sensitive elements due to scavenging by Iron (Fe) and Manganese (Mn) oxyhydroxides. These oxyhydroxides are soluble in their reduced states and insoluble under the oxic conditions that are found near the surface sediments. To evaluate changes in redox sensitive elements, we normalized their concentration to Fe. In addition, to evaluate changes in the source mineral contributions we normalize concentratons of Ca, Lanthinum (La), Aluminium (Al) to the conservative element Titanium (Ti).

Diatom Analysis

Sediments were prepared using 5 mg of dry sediment digested in 30% hydrogen peroxide solution and placed in a hot water bath for 2 days. Sediments were rinsed with de-ionized water and centrifuged, repeating until the solution measured neutral pH (Battarbee et al. 2001). Samples were prepared using Battarbee trays to allow calculation of absolute diatom numbers (Battarbee et al. 2001). These diatom slurries were pipetted onto glass cover slips, dried overnight and mounted using Naphrax mounting

medium. Alternate sediment sections were chosen for enumeration with a minimum of 500 diatoms for each slide. Diatoms were counted using an Olympus BX51 microscope equipped with differential interference contrast optics and a 1.3 NA 100X oil-immersion objective.

Because sediment phosphorus profiles are not always representative of P loading rates (Carignan and Flett 1981), we use the diatom calibration set from Brahney et al. (2014a) to derive diatom-inferred dissolved phosphorus records for both North Lake sediment cores (NOR09-2, NOR08-2). Brahney et al. (2014a) provide a detailed description of the analysis. Briefly, they used only alpine and subalpine lakes to remove confounding differences in species composition that may arise from other gradients, for example, elevation and vegetation gradients (i.e. shallow grassland to piedmont lakes to alpine lakes). The constrained analysis indicated that total dissolved phosphorus contributed 25.6% of the explanatory power in the full model, and 12% of the variation while accounting for other variables. The full model was significant at $p<0.005$. In this analysis, both sediment phosphorus and pH were closely aligned to axis 1, suggesting these factors are similarly controlled and play a strong role in species composition (Brahney et al. 2014a).

Based on the ordination results presented in Brahney et al. (2014a) , we are confident that a Weighted Average Partial Least Squares (WA-PLS) regression (Ter Braak and Juggins 1993) can be used to reconstruct TDP concentrations in North Lake. A WA-PLS with leave-one-out cross-validation was used to develop the diatom-phosphorus transfer function using C2 software version 1.6.5 (Juggins 2005). We evaluated the optimum number of WA-PLS components based on the root mean squared error of prediction (RMSEp) using the leave-one-out cross validation. The optimum number of components was determined to be 2. We removed obvious outliers from the data set by plotting the observed versus the predicted values. Watershed and nutrient properties were normalized to the maximum value and scaled to unit variance. Diatom data were expressed as the proportion of species to the total count in each sample. Diatom species were limited to those that made up more than 2% in abundance and were found in more

than 5% of the lakes. To evaluate the predictive ability of the models, we used the root mean squared error and the correlation to the ordination axis.

Nitrogen deposition history

We use nitrogen isotope data to evaluate the historical contributions of atmospherically derived nitrogen to the North and Lonesome lake basins. Because atmospheric nitrogen is frequently isotopically lighter than the dissolved inorganic nitrogen (DIN) in aquatic habitats, an upcore decrease in $\delta^{15}N$ is often interpreted as an increase in the contribution of anthropogenic atmospheric nitrogen to the DIN pool (Holtgrieve et al. ; Wolfe et al. 2003). Nitrogen isotopes from the Fremont Glacier, Wind River Range, indicate that the $\delta^{15}N$ of N deposited in the region is predominantly negative and ranges from -3.15 to -5.88‰ ±0.27 (Naftz et al. 2011). A one to two per mil decrease in $\delta^{15}N$ occurred through the 1950-1990 period. An additional $\delta^{15}N$ value of -4.0‰ was obtained in 2000 from precipitation samples collected by the National Atmospheric Deposition Program at the Pinedale station (Figure 1, WY06). During diagenesis, the ammonification of organic nitrogen has the potential to alter the $\delta^{15}N$ isotope profiles (Bada et al. 1989; Lehmann et al. 2002). As such, we apply a diagenesis correction to the bulk dataset as defined in Brahney et al. (2014b). Sediments were freeze-dried, but not otherwise treated prior to N isotope analysis. Sediments were analyzed for carbon and nitrogen elemental concentrations and isotopic composition with a Finnigan Mat Delta Plus mass spectrometer at the University of Regina. Replicate samples indicated variation up to 0.02‰.

Sediment Pigments

Sediments from NOR08-2, NOR09-2, LSM08-2, and LSM09-2 were analyzed for pigments using methods outlined in Leavitt and Hodgson (2001) at the University of Regina. Cores were stored in opaque containers and kept cool until delivery to the University of Regina. Because Pheophytin-a is a

decomposition product of Chlorophyll-a, we use the ratio of Chlorophyll-a to Pheophytin-a as an index of pigment decomposition. Lutein, an indicator of chlorophytes, and Zeaxanthin, an indicator of cyanobacteria, are not easily differentiated based on their spectral signatures and are thus grouped together.

Evaluating potential controls on dust emissions

To evaluate the relationship between climate and dust deposition, we obtained historical climate data including precipitation and temperature from the National Oceanic and Atmospheric Administration's (NOAA) National Climate Data Center for Pinedale, Wyoming (Figure 1)(Peterson and Vose 1997). The Palmer Drought Severity Index (PDSI) and wind speed data for the southwestern Wyoming region (Sublette County) were obtained from the NOAA Western Region Climate Center and the National Climate Data Center.

To evaluate the potential role of human land-use on soil disturbance, we related dust deposition data to historical statistics on land-use upwind in southwestern Wyoming. Oil and gas operations are the main industrial activity in the Green River Basin. Fugitive dust is emitted during the construction of well pads, drilling, and throughout the lifetime of the well. In the short term, total suspended particulate (TSP) emissions are estimated to be 6.5 tons $well^{-1}$ yr^{-1} during the construction and drill phase. In the long-term, production emissions are estimated to be 14 kg $well^{-1}$ yr^{-1} (Blm 2008); therefore, both the number of wells drilled in a year and the total wells in existence can influence the amount of dust and other emissions generated. Using this information, we calculated the annual emission rate of dust based on estimates from the construction, drilling, and production phase of a well's lifetime. We estimated potential emissions from Trona mining based on the annual production in metric tonnes (Blm 2014).

Results

Dating

Radiocarbon age models for NOR08-2 are based on 8 radiocarbon ages, one $^{239+240}Pu$ age that marks the 1963 maximum activity peak (Ketterer et al. 2004), and the surface sediment coring date. Based on the CLAM spline age-depth relationship, the 95% confidence interval varies between ± 10 years near the surface of the core, to a maximum of ± 97 years through the rest of the core (Figure 3). No age reversals were found in the ^{210}Pb CRS age model for NOR09-2. The standard deviation increases from 0 (surface coring date) to ±26 years at 9 cm depth (Figure 4). Mass accumulation rates show increases through the last half century in both the percussion (NOR08-2) and gravity (NOR09-2) cores from North Lake (Figure 3 and 4). Mass accumulation rates increase from 0.02 to 0.05 g cm^{-2} yr^{-1} during this period. The early history of NOR08-2 core shows a decline in sedimentation rates to relatively stable conditions of 0.02 to 0.01 g cm^{-2} yr^{-1}. The peak early in the cores corresponds to a light-colored, coarser-grained interval composed mostly of mineral grains that likely represent a landslide in the catchment.

The radiocarbon age model for LSM08-2 is based on 4 radiocarbon ages. Since we were not able to ascribe a coring date to the surface sediments, the uncertainty in the age model increases to the surface off the core to a maximum of ± 113 (Figure 5). LSM09-2 showed no age reversals based on the ^{210}Pb activity, with a standard deviation increasing rapidly from 0 (surface coring date) to ± 59 years at 8 cm depth (Figure 6). The Lonesome Lake percussion core (LSM08-2) shows very little change in mass accumulation rates (Figure 4); however, the gravity core (LSM09-2) shows abrupt and large shifts in sedimentation rates throughout the core, most notably a decline from 0.07 to 0.02 g cm^{-2} yr^{-1} around the turn of the century. This is followed by a smaller increase in 1970 to 0.04 g cm^{-2} yr^{-1} (Figure 6). Lonesome Lake is situated in a basin below several cirque lakes and inlet streams; it is possible that sedimentation rate changes are tied to variations in inflow from these diverse sources.

Metal and nutrient profiles

The North Lake percussion core (NOR08-2) shows increases in some metals beginning between 14 to 17 cm depth (200-300 Cal.Yr. BP) including, P:Fe, Ca:Ti, Cu:Fe, as well as Fe normalized (Tin) Sn, Sr, Cobalt (Co), Chromium (Cr),Nickel (Ni), Zinc (Zn), and Lead (Pb). A more substantial increase in many elements occurs from 4 cm (~1960) (Figure 3). The North Lake gravity core (NOR-09) shows upcore increases in P:Fe, Ca:Ti, La:Ti, Al:Ti, P:Fe, Cu:Fe, Cr:Fe, Sn:Fe, and Ni:Fe (Figure 4). Certain heavy metals begin to decrease around 2.5 cm (~1985-1988), including Arsenic (As), Cadmium (Cd), Cr, and Antimony (Sb), and further decreases in Cu and Pb from 0.75 cm (~2007-2008), which may be a result of the recent US Environmental Protection Agency regulations regarding the emplacement of scrubbers in power plants and removal of lead from gasoline. A peak in most elements in the mid 1980's is likely due to a large fire that burned approximately 37,300 acres on the western slope of the Wind River Range (*personal communication, Eric Ege, USFS*). Geochemical changes in LSM08-2 are restricted to slight increases in P:Fe, Ca:Ti, with very little change in other elements and their ratios (Figure 5). Surface sediment geochemical profiles indicated that NOR09-2 has 1.3 to 6x times the concentration of elements associated with atmospheric deposition than LSM09-2, including P, Pb, Cu, Cr, Cd, and Sb (Brahney et al. 2014a) .

Dust deposition in North Lake

The $^{147}Sm/^{144}Nd$ and ε_{Nd} values show that our dust samples are similar to those measured for other North American dusts, and the bedrock data fall within the range measured for Archean rocks (Grousset and Biscaye 2005; Neff et al. 2008)(Figure 7). Most of the Lonesome Lake sediments are statistically indistinguishable from the bedrock values, whereas North Lake sediments plot closer to the measured dust values and many are statistically distinct from the bedrock values (ε_{Nd} F=23.83, p<0.001). North Lake sediment core isotope profiles show a recent (~1940) shift in isotope composition towards the dust

isotopic composition (Figure 3). Both the mixing model and the mass-fraction calculations indicate similar catchment-integrated fluxes of dust (Table 3). Both analyses indicated that dust fractions rose from less than 5% prior to the 1940's, up to ~20 % in the latter half of the 20th century. From 1940 to 2006 the sedimentation rate of the mineral fraction increased from 137 g m^2 yr^{-1} to 299 g m^2 yr^{-1} and the dust flux increased from less than 1 to 53-68 g m^{-2} yr^{-1}, accounting for 33-42% of the rise in the mineral flux (Table 3). A small increase in the catchment-integrated flux to the sediments was also observed at approximately 1070 Cal. Yr. BP (Table 3). The surface sediment in LSM09-2 was the only interval that displayed a shift towards isotopic values of dust (Figure 5). Calculations as above indicate approximately 10% of this interval is exogenous dust material.

Diatoms

Diatom species composition

Diatom frustules remained well-preserved through the length of the cores and we found no evidence of dissolution. Several changes in diatom species composition and abundance have occurred over the last 2000 years. Both gravity and percussion cores from North Lake reveal a recent and large increase in diatom abundance (Figures 3 and 4). The inception of the recent upswing occurred sometime between 1935 and 1950, culminating in 2009 at the highest diatom abundances observed in the last 2000 years. Previous increases in abundance occurred around 150-250 Cal. Yr. BP (10-15 cm) and 1070 Cal. Yr. BP (45 cm). Similarly, the Lonesome Lake gravity core (LSM09-2) shows an increase in abundance towards the surface sediments (Figure 6) and the percussion core (LSM08-2) shows a large increase in diatom abundance around 1100 Cal. Yr. BP (Figure 5).

The changes in species composition observed recently are distinctly different than the changes in species composition that occurred in the previous increases in diatom abundance, indicating potentially different causes for the modern and historical shifts in diatom production. In the North Lake cores we see recent increases in *Asterionella formosa, Pseudostaurosira brevistriata, Staurosira venter*, and *Fragilaria*

capucina var *gracilis*, with concurrent decreases in *Discostella stelligera*, *Aulacoseira alpigena*, and most *Achnanthidium* species. In the Lonesome Lake cores we see recent increases in *A. formosa*, and *P. brevistriata*, with lower levels of *D. stelligera* and *Achnanthidium* species in the surface sediments (Figure 4). The shifts in Lonesome Lake are similar to those in North Lake, though the magnitude of shift is substantially smaller, for example *A. formosa* concentrations in the Lonesome Lake gravity core (LSM09-2) surface sediments are still fairly low at ~3-5% (Figure 6). The calibration diatom set from Brahney et al. (2014a) suggests that this shift in species composition is limited to these and other lakes in southwestern region of the Wind River Range, the surface sediments of other lakes in the Northwestern, and Southeastern region of the range either have low *A. formosa* concentrations (<2%) or the species is not present at all, whereas *D. stelligera, Aulacoseira* sp*, Achnanthidium spp*, and small fragilarioid species dominate.

Diatom TDP Transfer Function

Total phosphorus reconstructions in both the percussion and gravity cores for North Lake show strong upward increases in TDP concentration (Figures 3 and 8). The model fit between the observed TDP and the model estimates of TDP gives an r^2 of 0.91, an RMSEp of 2.05, and the t-test *p*-value is 0.033. The percussion core indicates that concentrations between 3 and 4 $\mu g\ L^{-1}$ persisted for much of the last 2000 years, until a recent rise to approximately 9 $\mu g\ L^{-1}$ occurred. The gravity core shows a similar history, and indicates that the increase in TDP concentration occurred sometime between 1900 and 1940 to the present measured value averaging 12 $\mu g\ L^{-1}$ (Figure 8).

Sediment Carbon and Nitrogen

Percent carbon, nitrogen, and isotopic compositions of organic matter in the gravity cores show a general pattern that one might expect as a result of diagenesis, specifically a first-order exponential decrease in %C and %N downcore with a corresponding increase in the C:N ratio (Colombo et al. 1996;

Galman et al. 2008). These profiles tend to occur from diagenesis because nitrogen-rich proteins from algal material have a tendency to degrade faster than carbon-rich structural compounds (Galman et al. 2008). Given that the core profiles of the C:N ratio, %C and %N suggest diagenesis, we felt it was appropriate to correct the data for diagenetic alteration using the model created by Brahney et al. (2014b). Prior to diagenesis correction, the $\delta^{15}N$ profiles in North Lake show depletion beginning around 1926 by approximately 1.8‰, and in Lonesome Lake the depletion begins around 1960 with an upcore offset of 3.2‰. After applying the diagenesis model, nitrogen isotope concentrations in both Lonesome and North Lake show the effects of anthropogenic nitrogen deposition beginning around 1960 with an isotopic depletion between 1 and 2‰ (Figure 4 and 6).

Pigments

Increases in specific pigments in the North Lake gravity core (NOR09-2) occur at different times in the core profile (Figure 4). The ratio of Chlorophyll-a to Pheophytin-a degrades rapidly showing a first-order exponential decay. Most other pigments in the core do not mimic this profile. Beginning prior to 1974, Echinenone, Canthanthaxin, Diatoxanthin, and Lutein+Zeaxanthin increase, while Alloxanthin decreases in concentration upcore. Zeaxanthin, Canthanthaxin, and Echinenone are all indicators of cyanobacteria, and Alloxanthin is an indicator of cryptophytes. Lonesome Lake (LSM09-2) shows abrupt upcore increases in all pigments beginning around 1985 (2.25 cm) with no discernable changes in species composition (Figure 6). Note that up until 2.25 cm the profiles parallel the ratio of Chlorophyll-a to Pheophytin-a, after which some similarities can be drawn between these profiles and that of the N deposition record. Both percussion cores (NOR08-2, LSM08-2) show increases in the Chlorophyll-a to Pheophytin-a ratio ~1100 Cal. Yr. BP, indicating an increase in preservation and/or productivity at this time (Figures 3 and 5).

Climate and anthropogenic controls on dust emissions

Precipitation data for Pinedale, Wyoming, shows an increasing trend through the last century (p<0.1) while temperature shows no significant trend. Wind statistics were only available post 1999 for the Green River Valley region, and show a rise in the number of wind events greater than 10 m s^{-1} in the last few years of the decade. The PDSI for the southwestern Wyoming does not indicate a drying trend. Negative values (drier years) existed from 1932-1935, 1940-1941,1953-1964, 1975-1977m 1989- 1991, 20001-2004, and 2007-2009. There were no significant relationship between sediment dust profiles and precipitation (r^2=.007, p =0.75), wind (r^2=0.18, p =0.41), or PDSI (r^2=.01, p=.71).

Calculated TSP emissions from the oil and gas industry begin in 1923 when the first well was drilled. TSP rises to over 100 metric tonnes per year (mta) in 1940 and culminates to an initial peak of a 1021 mta in 1963. TSP then drops to between 500-800 mta before beginning to rise in the late 1980's and peaking in 2008 at 6589 mta (Figure 8). Trona mining began in 1950 and production rose rapidly through the 1960's, 70's, and 80's eventually stabilizing at just under $2x10^7$ mta (Figure 8). Strong significant relationships exist between calculated TSP emission (log) and the dust flux found in the sediment profiles (mass fraction; r^2= 0.79, p<0.0001, constrained-least squares; r^2=0.63, p<0.0001). Similar strong relationships were found between the production of Trona in the Green River Valley (log) to dust flux in the sediments (mass fraction; r^2= 0.45, p<0.005, constrained-least squares; r^2=0.57, p<0.005).

Discussion

Sediment core analyses from Wind River lakes reveal the presence of atmospheric nitrogen in both North and Lonesome Lakes (Figures 4 and 6) and the presence of exogenous dust in North Lake from ~1940, and in the surface sediments of Lonesome Lake. The degree of ecosystem change was greater in North Lake than in Lonesome Lake and parallel the dust concentrations in lake sediments, suggesting that the main driver of change in the southwestern alpine lakes of the Wind River Range is an increase in dust deposition, or perhaps the synergistic effects of nitrogen and dust deposition.

The variation of $^{143}Nd/^{144}Nd$ and $^{87}Sr/^{86}Sr$ isotope data between the bedrock and dust material in the North Lake sediments allowed separation of exogenous dust material from the North Lake basin. Based on this analysis, we found large increases in the dust beginning ~1940 and rising from near zero to a catchment integrated flux of 50-70 g m^{-2} yr^{-1} peaking around 2006 (Table 3, Figure 8). Geochemistry data support this conclusion with changes in the elemental chemistry of immobile elements from ~1950 (Figure 3 and 4). Calcium, normalized to Ti, and P, normalized to Fe, both show increases in sediment concentration around the same time, both of which are markers of dust deposition in the region (Brahney et al. 2013; Brahney et al. 2014a). The dust fractions within the sediments explain a large fraction (33-42%) of the increase in sedimentation rates that occurred through this time period, the remaining fraction likely due to the recorded increase in precipitation.

Comparing the sediment derived dust flux between the two lakes, we find that the surface sediments of Lonesome Lake indicate a sedimentation rate of 25.5 g m^{-2} yr^{-1}, compared to 58 g m^{-2} yr^{-1}in North Lake. Normalizing to the catchment area to lake area, we find that at peak deposition in the surface sediments, North Lake received 1.6x more dust than Lonesome Lake. This estimate is within range of those derived from the geochemical data, where elements associated with atmospheric deposition were 1.3 to 6 times greater in North Lake. Lower dust sedimentation rates in Lonesome Lake may be due to lower dust deposition rates arising from the protection of the continental divide, or because the catchment area to lake area ratio is twice as large for North Lake. A lower deposition rate, and larger lake volume may also account for the lower TDP values measured in Lonesome Lake.

A variety of potential factors can contribute to an increase in soil erosion and transport from the semi-arid Green River Valley upwind of the Wind River Range. Soil erosion rates are controlled by both erosion potential (wind speeds), and factors that increase the availability of soil for erosion. The latter is influenced by factors that decrease soil moisture (drought), and decrease the amount of soil-stabilizing vegetation (drought, human land-use) (Field 2009). An increase in wind speeds through the latter half of the 2000's can account for greater dust transportation through this time. However, we find no statistical

relationship between wind, precipitation, or the PDSI to dust deposition trends. In contrast, estimated TSP emissions from the oil and gas operations in the Pinedale Anticline show a remarkably similar profile to the dust flux in the North Lake sediments, suggesting a strong relationship between anthropogenic activity and dust deposition to the alpine lake (Figure 3 and 8).

Several proxies appear to change in the 19th century, prior to the recent onset of enhanced dust flux as determined by $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotope ratios, including the diatom inferred TDP concentrations (Figure 8), several diatom species, Ca:Ti, Al:Ti, and Cu:Fe, and Cr:Fe ratios, and Pb concentrations in the sediments (Figure 4).Though not documented by isotope analysis, it is possible that dust fluxes also increased with the development of non-indigenous anthropogenic activities (ranching, homesteading, mining), which date back to the 1860's in the Green River Area (Hausel 1994; Cassity 2011). Increases in dust deposition around that time would be consistent with results found in other dust studies in the inter-mountain west (Neff et al. 2008; Reynolds et al. 2010).

Implications of increased dust emissions

The historical intensity of industrial activities adjacent to the Wind River Range parallel the 1) calculated dust flux to the North Lake basin based on $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotopes, 2) sediment phosphorus concentrations, 3) the diatom inferred phosphorus concentration, and 4) the Ca:Ti ratios in sediments. The temporal consistency of these events strongly suggests that dust is contributing significantly to the phosphorus and calcium budgets of lakes in the southwestern portion of the range. The ordination analysis from Brahney et al. (2014a) indicate that lake phosphorus, pH, and alkalinity are similarly controlled across the region and play a strong role in diatom species composition. The WA-PLS regression reconstructs plausible historical phosphorus concentrations of 3 to 5 μg P L^{-1}, which are similar to non-dust affected lakes in the region (Brahney et al. 2014a). Further, the time-frame is roughly consistent with the timing of observed increased dust (and phosphorus) flux to the basin, and sediment phosphorus concentrations (Figures 3, 4, and 8). These independent lines of evidence strongly support the

hypothesis that dust is the primary cause for increased phosphorus to the basin over the last 50 to 100 years, and is a significant factor in changing species composition. From ~1940 TDP concentrations have more than doubled in North Lake and currently average 12 μg P L^{-1}. Because North Lake is remote, it is difficult to explain such a dramatic change in P concentration in the absence of an exogenous dust source.

Increases in phosphorus concentration generally lead to increases in lake productivity (Smith 1979). In the North Lake sediment cores an increase in the organic flux to the sediments is observed, as well as an increase in diatom productivity and a shift in plankton species composition, all of which begin circa 1940. Diatom cell counts indicated that diatom productivity increased tenfold, from approximately 1500 cells μg^{-1} sediment in ~1940 to 15000 cells μg^{-1} sediment in late 2000's (Figure 4). The relative abundances of *Pseudostaurosira brevistriata*, *Asterionella formosa, Fragilaria capucina* var. *gracilis*, and *Staurosira construens* var. *venter* increased, while the relative abundances of *Discostella stelligera,* small fragilarioid, and *Achnanthidium* species declined (Figure 3, 4 and 6). Both *A. formosa* and *P. brevistriata* are common to mesotrophic environments, and *A. formosa* in particular has been known to respond to phosphorus pollution (Reavie et al. 1995; Wolin and Stoermer 2005). The decline in small benthic fragilarioid species and *D. stelligera* is consistent with eutrophication as *D. stelligera* is commonly found in circumneutral oligotrophic lakes (Enache and Prairie 2002; Rühland et al. 2003). Similarly, small fragilarioid species are common in alpine and northern lakes because they can tolerate the short growing seasons and the duration of ice cover (Karst-Riddoch et al. 2009). The diatom species present in the early history of the core are similar to those found in lakes elsewhere in the range (Brahney et al. 2014a) indicating historically oligotrophic conditions in North Lake.

A. formosa appears to be in general a common disturbance species proliferating after a variety of environmental perturbations, including N deposition (Wolfe et al. 2003; Saros et al. 2005), temperature increases (Solovieva et al. 2008), as well as phosphorus loading (Reavie et al. 1995; Wolin and Stoermer 2005). In the Wind River Range temperature increases are not likely an important factor because both air and lake water temperature measurements in the range do not show upward trends though this time

period. Because *A. formosa* is a good competitor for P when N is in adequate supply (Saros et al. 2005; Michel et al. 2006), it is likely that both N and P are synergistically contributing to large increases in *A. formosa* productivity. However, because both North and Lonesome lakes receive similar amounts of nitrogen deposition and *A. formosa* is not present in high abundances (3-5%) in the Lonesome Lake gravity core record (Figure 6), phosphorus appears to be the main driver for the proliferation of *A. formosa* in North Lake. Further, the increase in *A. formosa* abundance through time in North Lake follows a similar profile to that of the dust flux increase; *A. formosa* concentrations show a small increase beginning sometime after 1930, which is then followed by a large increase in the late 2000's.

Sediment pigment profiles in North Lake show large upcore changes that can be interpreted as increases in productivity and shifts in species composition. Due to high degradation rates, pigment concentrations would be expected to increase upcore; however several other lines of evidence (sedimentation rates, absolute diatom counts) also indicate increases in productivity. The changes in pigment concentrations upcore indicate an increase in cyanobacteria and a decrease on cryptophyte populations, providing some insight into potential algal species changes that may have occurred in the recent history of North Lake. Cryptophytes are often characteristic of oligotrophic waters because they utilize a variety of strategies, including mixotrophy, which can enable them to utilize two different feeding strategies when nutrients are low (Jansson et al. 1996; Kalff 2001). A decline in cryptophytes may be due to a lower ability to compete with other algal species at higher phosphorus levels as cryptophytes have high N requirements (Jansson et al. 1996). Cyanobacteria, while found in diverse habitats both as planktonic and benthic species, generally have higher temperature and phosphorus requirements (Pick and Lean 1987), thus an increase in cyanobacteria presence may be due to the increase in phosphorus availability and a rise in pH due to carbonate deposition. Though diatom populations have also increased, the pigments suggest that diatoxanthin did not increase to the same degree as the similar, or more chemically stable, carotenoids from cyanobacteria, including Canthanthaxin, Echinenone, and Lutein+Zeaxanthin. This suggests that there has been a disproportionate increase in cyanobacteria with

respect to diatoms due to dust increases. Cyanobacteria may respond more favorably to dust (carbonate) deposition due their ability to exploit HCO_3^- at higher pH's when other algae may become DIC-limited (Badger and Price 2003).

Both diatom and pigment proxy data indicate substantial and recent changes in the plankton species community. Of some importance, the data indicate increases in cyanobacteria and *A. formosa* populations in North Lake, both possibly due to increased phosphorus and/or carbonate loading. These particular shifts in plankton populations can have negative consequences for higher trophic levels. Cyanobacteria and *A. formosa* are both low quality food items for zooplankton that result in low fecundity and diminished growth rates (Balseiro 1991; Wilson 2006).

Biological indicators from the sediment cores show signs that lakes in the area were affected by the widespread drought that occurred around 1100 Cal. Yr. BP (Woodhouse and Overpeck 1998). During this time interval, an increase in production occurs but is not accompanied by a change in species composition. Lake level lowering due to drought can lead to an increase in productivity through increases in the concentrations of dissolved constituents, reworking of nutrients and sediments form near shore environments, as well as a change in benthic habitat area. At this time interval, a small increase in dust, and presumably nutrient, deposition occurred. However, the magnitude of the increase is dwarfed by recent increases in the dust flux, and did not occur with simultaneous increases in nitrogen deposition. These differences may account for the recent changes in species composition that are absent from the increase in productivity around 1100 Cal. Yr. BP. The data strongly suggest that upwind industrial activities in the Pinedale Anticline contribute to the increase in dust flux.

Although the recent changes to dust and nitrogen deposition are still relatively modest (3.3 g m^{-2} yr^{-1} dust, 1.5 kg ha^{-1} N), the biogeochemical and biological implications are not. During the period from 1940 to 2008, North Lake underwent a shift from oligotrophic conditions (3-5 μg P L^{-1}) to mesotrophic conditions (12-14 μg P L^{-1}). In addition to having higher dissolved phosphorus concentrations, North Lake had 5 times the particulate concentration of phosphorus than lakes in other regions (Brahney et al.

2014a). These changes are associated with broad shifts in plankton communities from compositions similar to regions elsewhere in the range to distinct populations with lower food quality value. Dust and nitrogen deposition in general compromise water quality through increased nutrient and heavy metal concentrations. Further, the observed shift in algal species composition and decline in diversity affects food quality for species at higher trophic levels.

A growing number of studies are showing the effects of atmospheric deposition on alpine aquatic habitats. Data from this study and other lake sediment studies suggest that in the western USA there are large recent increases in dust deposition to lake ecosystems (Neff et al. 2008; Reynolds et al. 2010), and that increases in dust flux can broadly influence mountain lake biogeochemistry (Morales-Baquero et al. 2006; Ballantyne et al. 2010). Because dust often contains appreciable amounts of phosphorus (Lawrence and Neff 2009) there may be substantive increases to the phosphorus nutrient budget in many alpine regions that receive dust. At present, there are no national networks capable of capturing large scale changes in dust emissions. This is because national atmospheric deposition networks do not measure particles above 10μm and desert dust is often made up of larger particles (Lawrence and Neff 2009; Neff et al. 2013). In areas near deserts, these larger particles can contribute 50% of the particle concentrations (Lundgren 1984). This means that dust phosphorus deposition may be unrecognized by current monitoring networks and thus an underappreciated cause of ecosystem changes to high elevation lakes in the western US. Our observations, combined with others across western North America and Southern Europe provide a consistent conceptual model by which arid ecosystems when disturbed by climate, humans, or both are vulnerable to widespread soil loss that can be transported to nearby alpine ecosystems supplying them with both nutrients and pollutants. For example, near the Saharan Desert, one of the largest dust source regions in the world, recent increases in dust emissions have been tied to regional commercial agriculture and climate change (Mulitza et al. 2010). Dust phosphorus additions to the Sierra Nevada Mountains in Spain has been implicated in controlling stoichiometric nutrient availability, stimulating bacterial abundance, and decreasing phytoplankton diversity (Pulido-Villena et al. 2008;

Reche 2009). In comparison to the Sahara, the Green River Valley is a small dust producing region, yet similar and substantial effects have occurred.

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Table

Table 1. Lake characteristics and nutrient concentrations for North and Lonesome lakes in the Wind River Range. Data reproduced from Brahney et al. 2014a.

Lake	Latitude	Longitude	Altitude (masl)	DIN ($\mu g L^{-1}$)	TDP ($\mu g L^{-1}$)	Part. P ($\mu g L^{-1}$)	Part. Ca ($mg L^{-1}$)	Diss. Ca ($mg L^{-1}$)	Catchment Area: Lake Area	Lake Area (acres)	Max Depth
North	42.76	-109.06	3085	234	12.5	5.43	8.3	3.5	59	7	9
Lonesome	42.78	-109.22	3101	162	4.4	0.25	3.3	2.5	32	34	15

Table 2. Radiocarbon and calibrated ages for Wind River Range lake sediment cores

USGS National Center Sample ID	Lake	Core/Depth	Material Dated	$\delta^{13}C$	^{14}C Age	Range ±	Cal. Yr. BP (2σ)
WW8008	Lonesome Lake, WY	LSM1-08/12.5-13	wood	-25	355	30	316-496
WW8241	Lonesome Lake, WY	LSM1-08/25-25.5	plant material	-25	450	30	472-535
WW8009	Lonesome Lake, WY	LSM1-08/26.5-27	wood	-25	605	35	543-655
WW7238	Lonesome Lake, WY	LSM1-08/33-33.5	wood	-23.43	1220	35	1061-1190
WW8242	Lonesome Lake, WY	LSM1-08/34-34.5	plant material	-24.01	960	35	791-933
WW8244	Lonesome Lake, WY	LSM2-08/8-8.5	wood	-23.13	670	25	562-673
WW7239	Lonesome Lake, WY	LSM2-08/27.5-28	wood	-24.11	945	30	793-926
WW7240	Lonesome Lake, WY	LSM2-08/58-58.5	wood	-22.75	1575	30	1399-1532
WW8243	Lonesome Lake, WY	LSM2-08/73.5-74	plant material	-25	1875	30	1726-1880
WW7241	Lonesome Lake, WY	LSM2-08/88.5-86	wood	-26.58	2065	35	1945-2128
WW7242	North Lake, WY	NOR1-08/8.5-9.5	wood	-23.26	335	30	304-474
WW8248	North Lake, WY	NOR1-08/11-11.5	plant material	-23.51	815	30	682-780
WW7243	North Lake, WY	NOR1-08/25-25.5	wood	-24.39	1140	30	978-1068
WW8245	North Lake, WY	NOR2-08/42-43	plant material	-22.02	1185	30	1051-1179
WW7244	North Lake, WY	NOR2-08/44.5-45	wood	-25	1155	35	978-1172
WW8246	North Lake, WY	NOR2-08/49-49.5	plant material	-23.57	1225	25	1068-1185
WW7245	North Lake, WY	NOR2-08/63.5-64	wood	-23.61	1570	30	1371-1528
WW8247	North Lake, WY	NOR2-08/71-71.5	plant material	-22.88	1715	25	1555-1697
WW7246	North Lake, WY	NOR2-08/87	wood	-25.28	1890	30	1729-1900

Table 3 Estimated flux of dust to the lake basin using Constrained-Least Squares analysis (CLS) and isotope Mass Fraction (MF) calculations for North Lake sediments (NOR08-2 & NOR09-2 combined).

Cal Yr. BP.	Core Depth	CLS $g\ m^{-2}\ yr^{-1}$	MF $^{87}Sr/^{86}Sr$ $g\ m^{-2}\ yr^{-1}$	MF $^{143}Nd/^{144}Nd$ $g\ m^{-2}\ yr^{-1}$	MF Average $g\ m^{-2}\ yr^{-1}$
-58	0.25	35	34	41	37
-56	0.5	68	69	37	53
-10	4.5	53	53	25	39
-3	5	16	0	5	2
7	5.5	8	9	0	4
54	8.5	1	0	0	0
94	9.5	1	0	7	3
123	10.5	1	0	0	0
290	15.5	0	0	0	0
778	31.5	0	1	0	1
1070	44.5	0	0	6	3
1628	72.5	0	0	5	3

Figure Legends

Figure 1. The Wind River Range in western Wyoming, cities, labelled in italics, and NADP sampling sites are shown using hollow circles (WYXX). Sample lakes are in the southern portion of the range on either side of the continental divide. North Lake (Lat: 42.756, Long: -109.207); Lonesome Lake (Lat: 42.778, Long: -119.216). Also shown is Black Joe Lake, where the deposition samplers were located.

Figure 2. Comparison of density plots between NOR08-1 and NOR08-2. Peaks in density that correspond to visual laminae were used to incorporate radiocarbon dates from NOR08-1 into the NOR08-2 age model. Correlation between density peak depths between cores was high at r^2=0.99, p<0.001. The location of the radiocarbon ages used from NOR08-1 are shown with black circles.

Figure 3. Sediment profile data for the North Lake percussion core (NOR08-2), including the CLAM age model and corresponding mass accumulation rates, geochemistry, calculated dust flux, representative diatom and pigment results and diatom inferred total dissolved phosphorus (TDP). Calculated dust flux is shown using both the mass fraction and constrained least squares (CLS) methods.

Figure 4. Sediment profile data for the North Lake gravity core (NOR09-2), including the ^{210}Pb constant rate of supply model (CRS) and associated mass accumulation rates, geochemistry, nitrogen isotopes, and representative diatom and pigment results.

Figure 5. Sediment profile data for the Lonesome Lake percussion core (LSM08-2), including CLAM age model and corresponding mass accumulation rates, geochemistry, and representative diatom and pigment results.

Figure 6. Sediment profile data for the Lonesome Lake gravity core (LSM09-2), including the ^{210}Pb constant rate of supply model (CRS) and associated mass accumulation rates, pigment, nitrogen isotopes, and representative diatom results.

Figure 7. εNd and ^{147}Sm/^{144}Nd isotope composition of bedrock, sediment, and dust for the North Lake catchment, southwestern Wind River Mountains. We include data from Frost et al. (1998) and Neff et al.(2008) for comparison. LLB refers to the Louise Lake Batholith, the underlying bedrock for both North and Lonesome Lakes.

Figure 8 Calculated TSP emissions and Trona production for the Pinedale Anticline shown alongside the calculated dust flux in North Lake sediments based on isotope mass fraction calculations (black line) and constrained least-squares analysis (grey line), and the diatom inferred total dissolved phosphorus concentrations in the North Lake short core (NOR09-2).

Figure 1

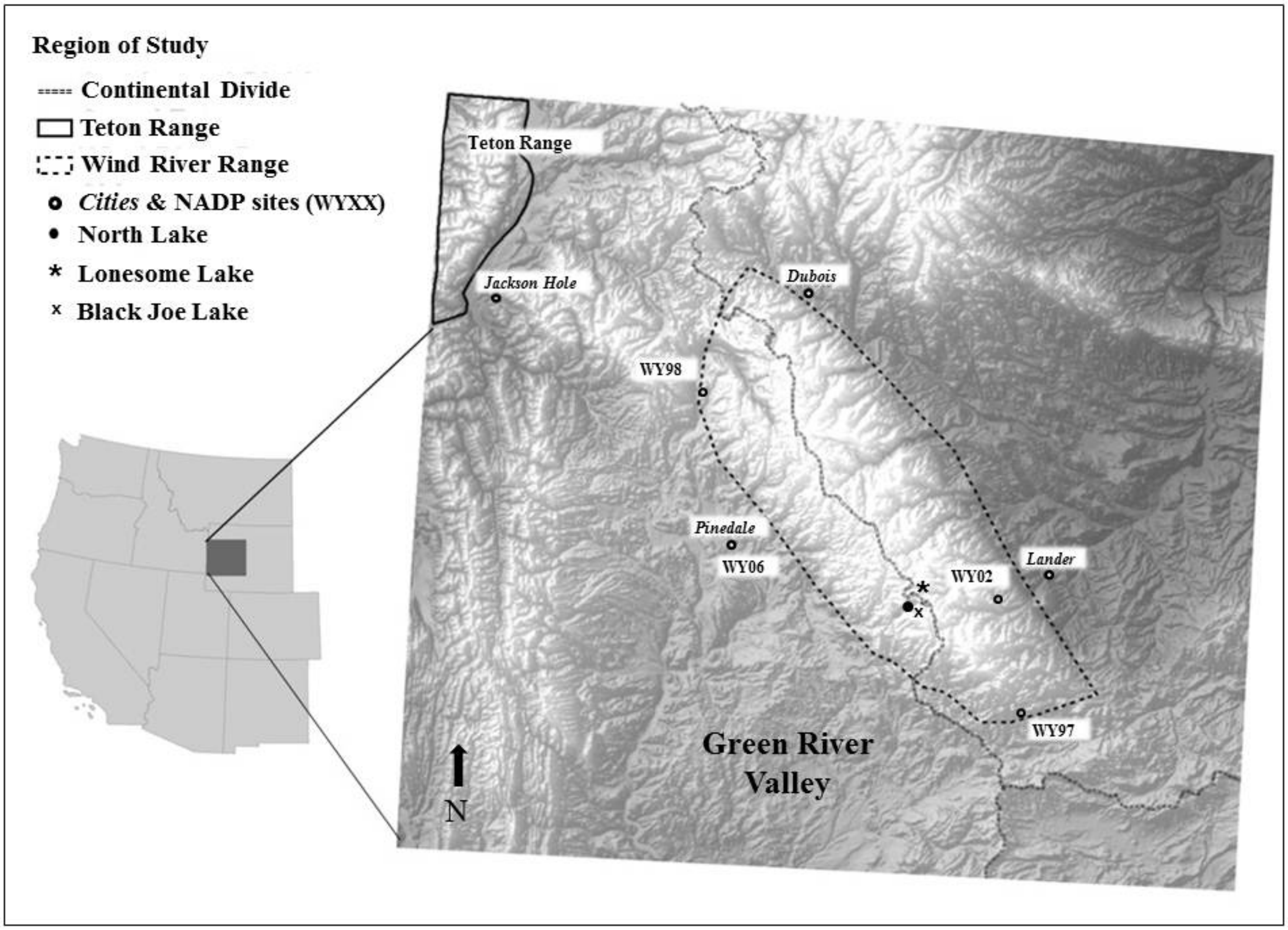
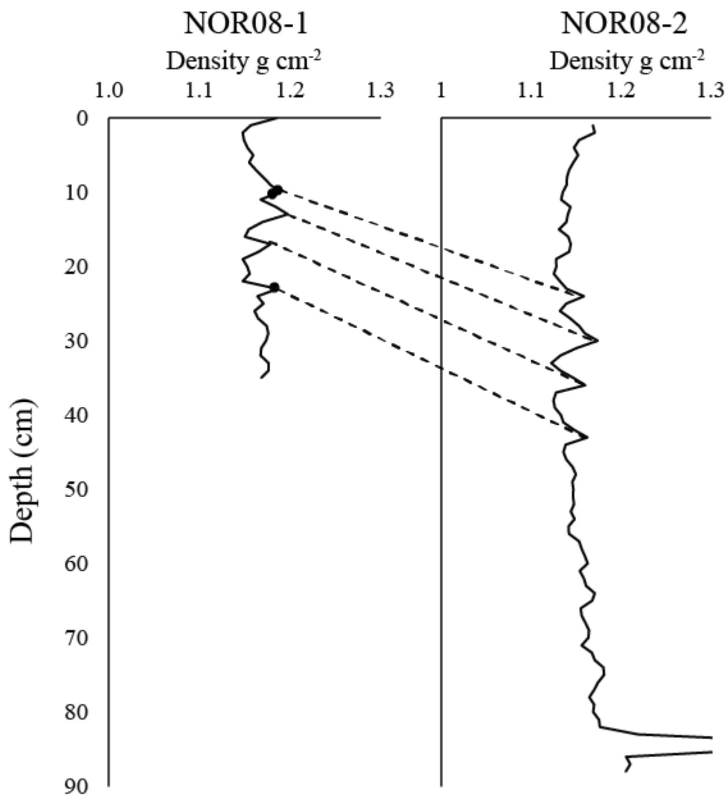

Region of Study
Continental Divide
Teton Range
Wind River Range
Cities & NADP sites (WYXX)
North Lake
Lonesome Lake
Black Joe Lake
Teton Range
Jackson Hole
Dubois
WY98
Pinedale
WY06
WY02
Lander
WY97
Green River Valley
N

Figure 2



NOR08-1
Density g cm-2
1.0
1.1
1.2
1.3
NOR08-2
Density g cm-2
1
1.1
1.2
1.3
Depth (cm)
0
10
20
30
40
50
60
70
80
90

Figure 3

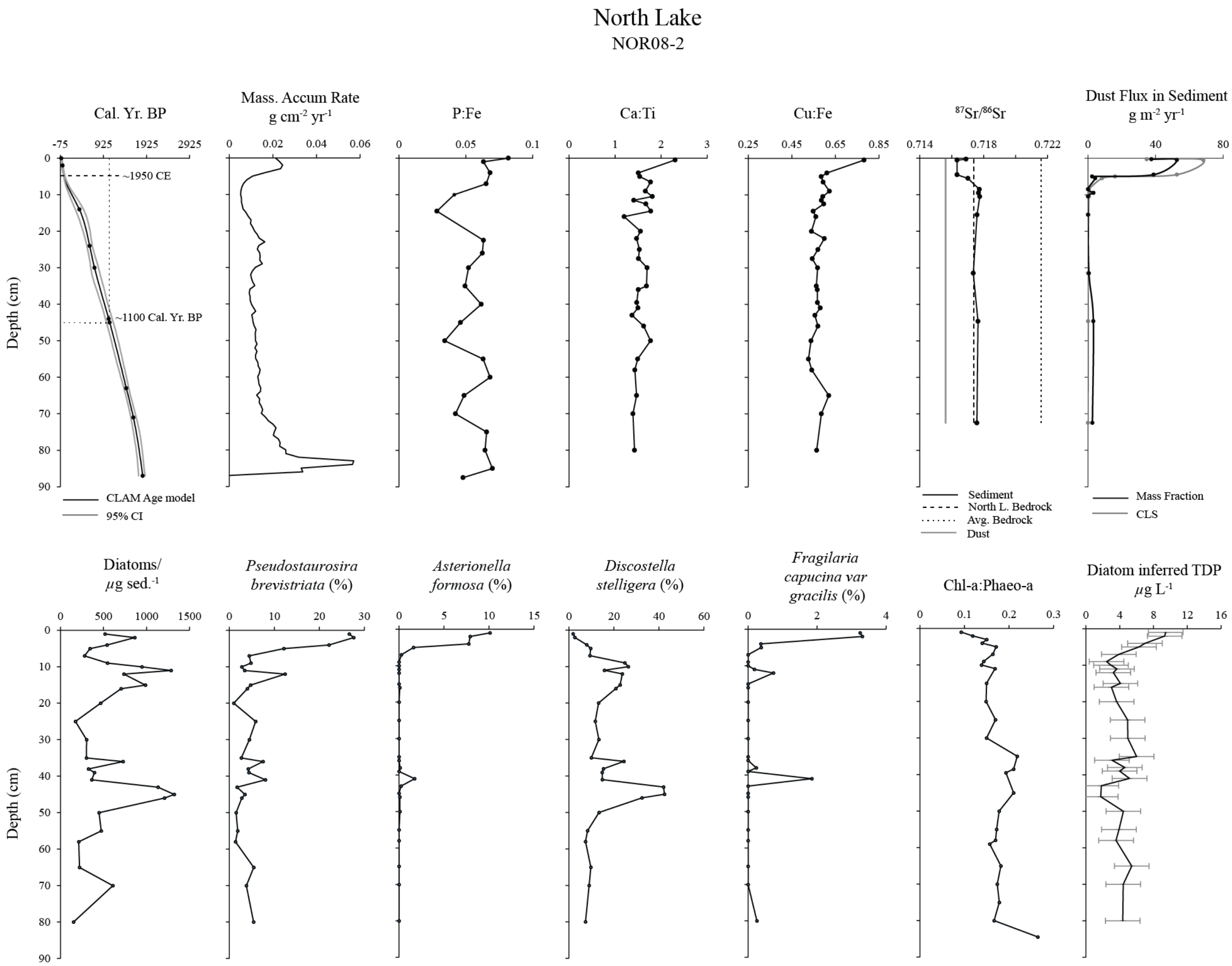

North Lake
NOR08-2
Cal. Yr. BP
Mass. Accum Rate
g cm-2 yr-1
P:Fe
Ca:Ti
Cu:Fe
87Sr/86Sr
Dust Flux in Sediment
g m-2 yr-1
Depth (cm)
~1950 CE
~1100 Cal. Yr. BP
CLAM Age model
95% CI
Sediment
North L. Bedrock
Avg. Bedrock
Dust
Mass Fraction
CLS
Diatoms/
μg sed.-1
Pseudostaurosira brevistriata (%)
Asterionella formosa (%)
Discostella stelligera (%)
Fragilaria capucina var gracilis (%)
Chl-a:Phaeo-a
Diatom inferred TDP
μg L-1

Figure 4

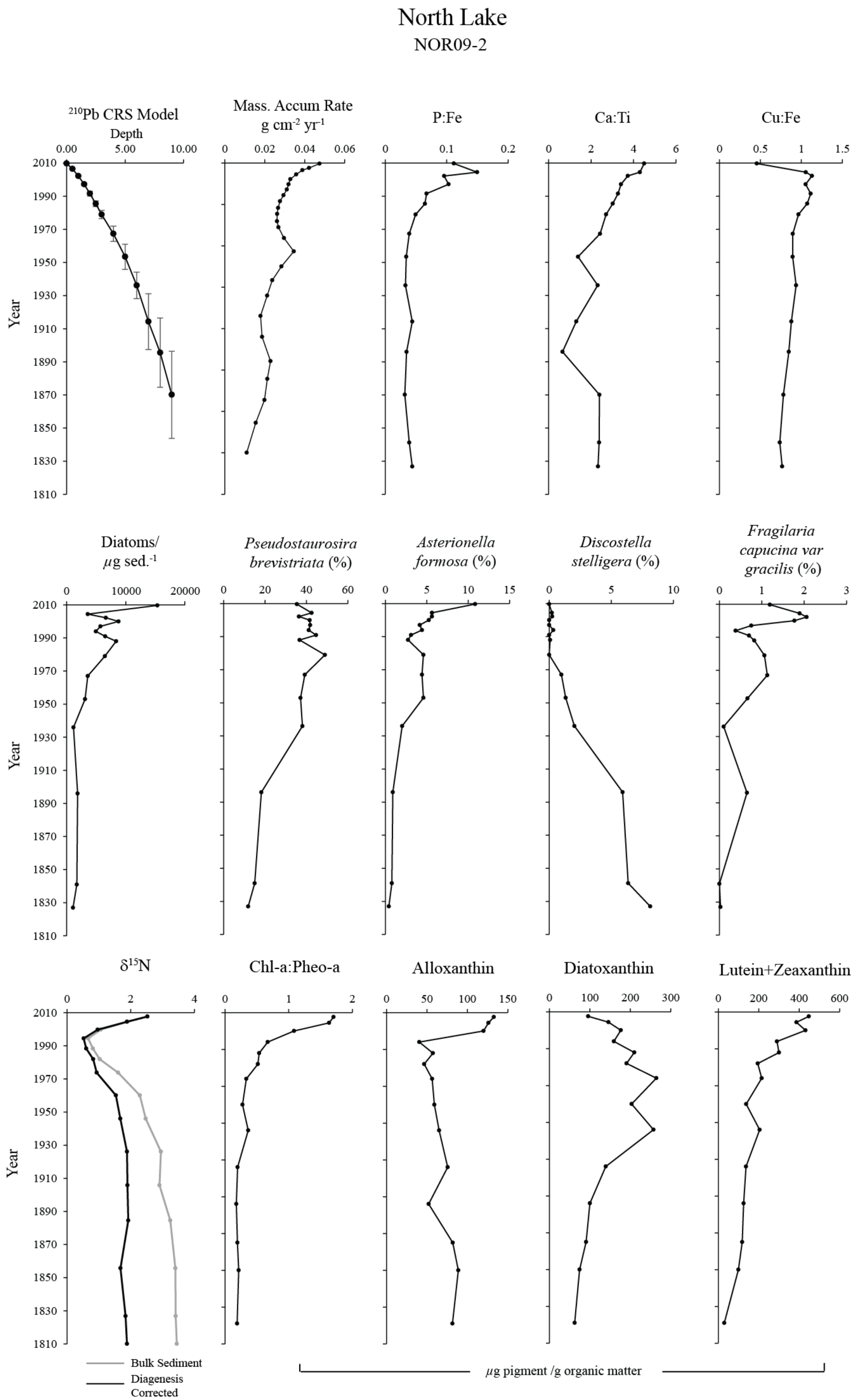

North Lake
NOR09-2
210Pb CRS Model
Depth
Mass. Accum Rate g cm-2 yr-1
P:Fe
Ca:Ti
Cu:Fe
Year
Diatoms/ µg sed.-1
Pseudostaurosira brevistriata (%)
Asterionella formosa (%)
Discostella stelligera (%)
Fragilaria capucina var gracilis (%)
δ15N
Chl-a:Pheo-a
Alloxanthin
Diatoxanthin
Lutein+Zeaxanthin
Bulk Sediment
Diagenesis Corrected
µg pigment /g organic matter

Figure 5

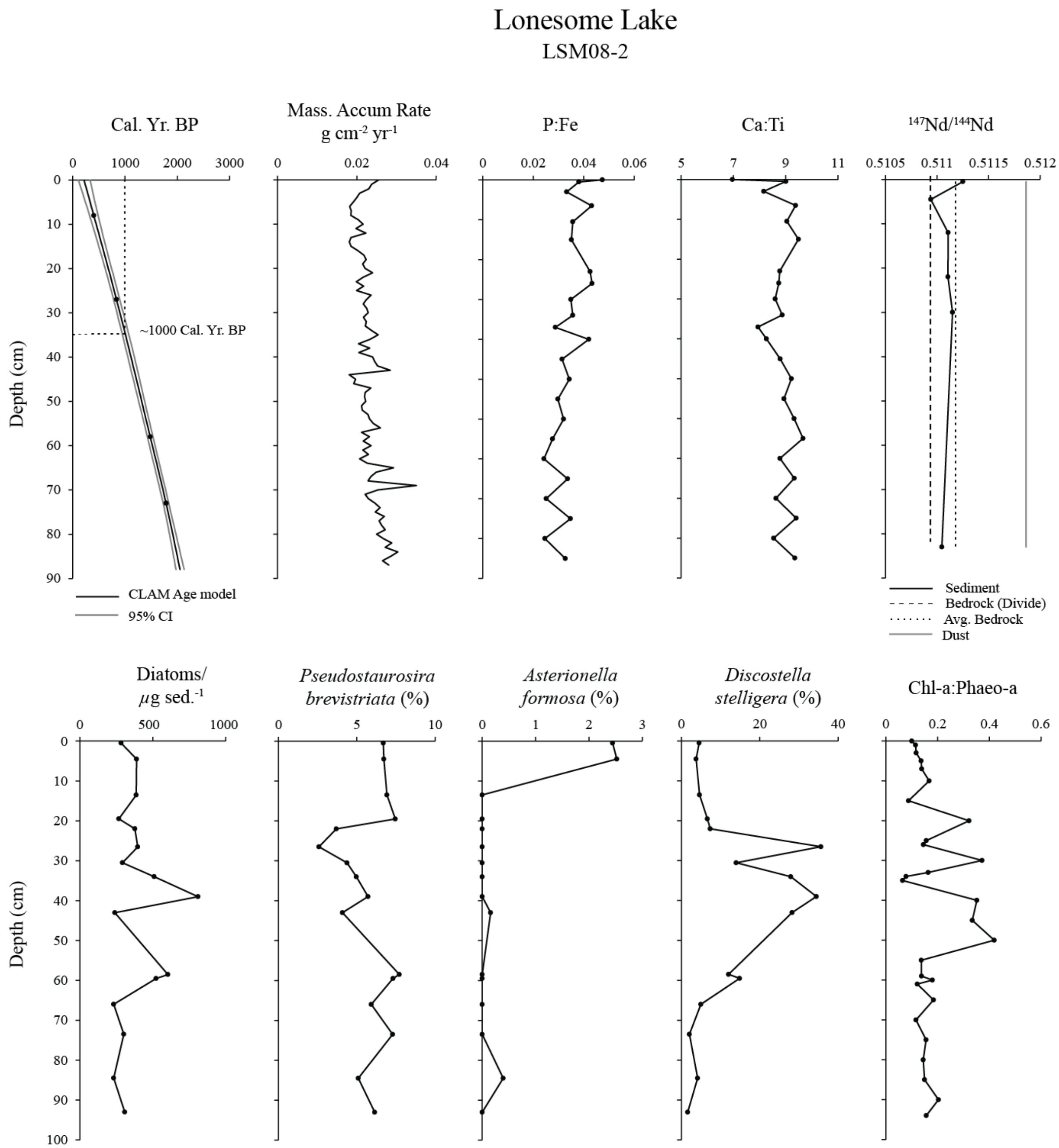

Lonesome Lake
LSM08-2
Cal. Yr. BP
Mass. Accum Rate g cm-2 yr-1
P:Fe
Ca:Ti
147Nd/144Nd
Depth (cm)
~1000 Cal. Yr. BP
CLAM Age model
95% CI
Sediment
Bedrock (Divide)
Avg. Bedrock
Dust
Diatoms/ μg sed.-1
Pseudostaurosira brevistriata (%)
Asterionella formosa (%)
Discostella stelligera (%)
Chl-a:Phaeo-a

Figure 6

Lonesome Lake
LSM09-2

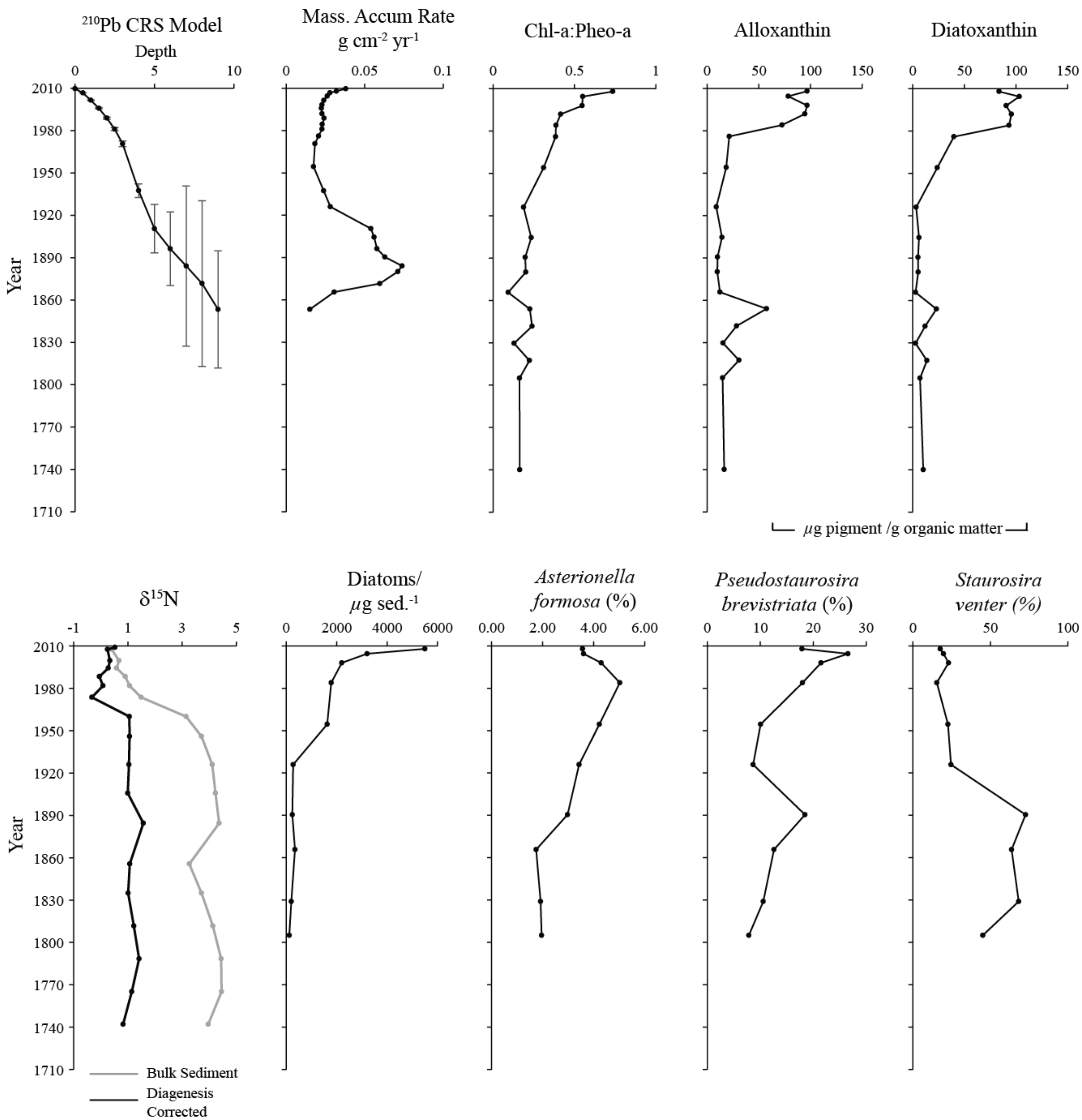


Figure 7

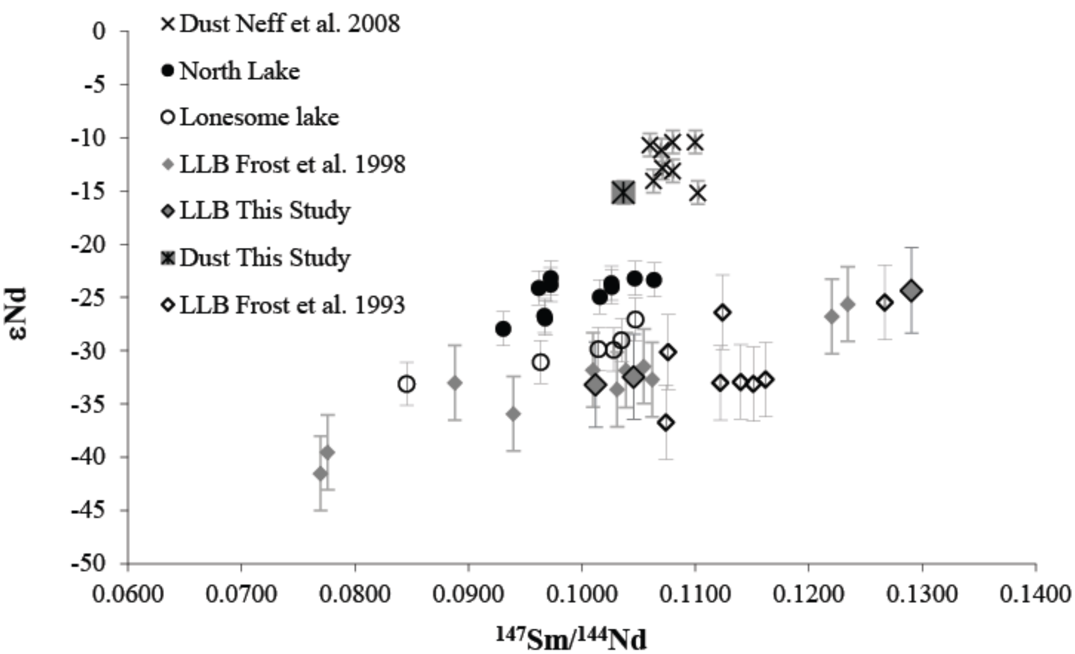
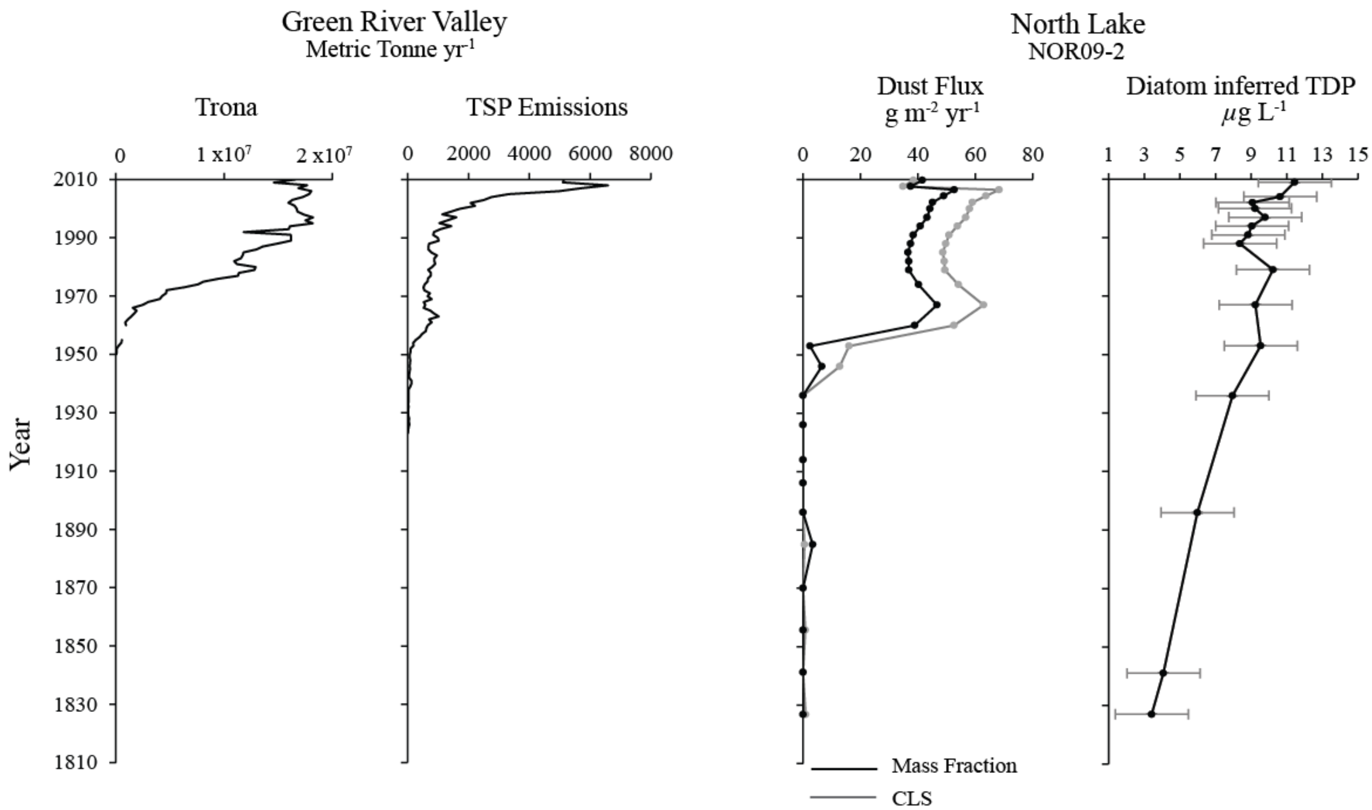

Dust Neff et al. 2008
North Lake
Lonesome lake
LLB Frost et al. 1998
LLB This Study
Dust This Study
LLB Frost et al. 1993
εNd
0
-5
-10
-15
-20
-25
-30
-35
-40
-45
-50
0.0600
0.0700
0.0800
0.0900
0.1000
0.1100
0.1200
0.1300
0.1400
$^{147}Sm/^{144}Nd$

Figure 8


Green River Valley
Metric Tonne yr-1
North Lake
NOR09-2
Trona
TSP Emissions
Dust Flux
g m-2 yr-1
Diatom inferred TDP
µg L-1
0
1 x10^7
2 x10^7
0
2000
4000
6000
8000
0
20
40
60
80
1
3
5
7
9
11
13
15
Year
2010
1990
1970
1950
1930
1910
1890
1870
1850
1830
1810
Mass Fraction
CLS