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Four decades of diluting phosphorus to maintain lake quality

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Abstract

Dilution with low-nutrient Columbia River Water (CRW) has markedly improved Moses Lake quality for 42 years. There were two phases of CRW volume input, which proportionately lowered total phosphorus (TP). Initially, spring–summer inputs averaged 130 × 106 m^3 during 1977–1988 reducing average TP from 152 to 65 μg/L in half the lake proximal to the inputs. That input represented 1.5 volumes of that half-lake volume. Inputs doubled through the mid-1990s, and nearly 2.5 times since 2000, decreasing TP to a 18-year average of 24 μg/L. Chlorophyll a (chl) decreased further from 18 μg/L during the early dilution years to about 6 μg/L as TP declined. Cyanobacteria biovolume declined to 57% of total biovolume during 1977–1988 from 98% before dilution. Less (65%) CRW water since 2016 led to higher TP (41 μg/L) and chl (18 μg/L) in 2018, while cyanobacteria averaged 87% of total biovolume. More TP and cyanobacteria in 2018 are attributed to more internal TP loading. Increased N:P ratios have possibly given more advantage to the non-N-fixing cyanobacteria *Microcystis*, which comprised 82% and 74% of maximum cyanobacteria biovolume in 2017 and 2018. © 2019 Water Environment Federation

Practitioner points

- Lake total phosphorus (TP) was reduced 57% in the 1970s–1980s by adding large volumes of low-nutrient Columbia River water (CRW).
- Total P was further reduced by 65% since 2000 by more than doubling the earlier CRW input to an average spring–summer concentration of 24 μg/L.
- Less (65%) CRW during 2017–2018 led to higher lake TP (41 μg/L) and a worse cyanobacteria bloom in 2018.
- *Microcystis*, an non-nitrogen fixer, was the dominant cyanobacteria in 2017–2018 likely related to higher N:P ratios.

Key words

dilution; N:P ratio; phosphorus; trophic state

Introduction

Addition of large volumes of low-nutrient water to dilute the nutrient content of eutrophic lakes has been a seldom-used restoration method due to lack of source water (Cooke, Welch, Peterson, & Nichols, 2005). However, Moses Lake, in eastern Washington proximal to the Columbia River irrigation project, is supplied by large volumes of low-nutrient Columbia River water (CRW). Moses Lake was hypereutrophic during the 1960s to the mid-1970s, before the addition of CRW (Welch, Barbiero, Bouchard, & Jones, 1992). The lake was highly degraded with total phosphorus (TP) and chlorophyll (chl) averaging 152 and 58 μg/L, respectively, well above the eutrophic–hypereutrophic boundaries of 100 and 30 μg/L (Nürnberg, 1996). Harmful algal blooms (HABs) of cyanobacteria, *Aphanizomenon* and *Microcystis*, occurred throughout the summer, forming unsightly surface scums and were probably toxic. Although growth of algae requires nitrogen in larger quantities than phosphorus, the latter is the key nutrient that drives eutrophication of lakes worldwide and Moses Lake was not an exception (Welch, 2009).

The potential benefit of adding CRW to the lake was recognized in the early 1960s because CRW was low in phosphorus and nitrogen, and the infrastructure was in place to route CRW from the East Low Canal (ELC) through Rocky Coulee Wasteway (RCW) and Crab Creek (CC) into Parker Horn and through the South Lake outlet to irrigators downstream (Figure 1). Transporting large quantities of CRW with a low TP concentration of 20 μg/L would dilute the high TP lake water and proportionately reduce the concentration of algae. There was precedent for using RCW as a feed route for irrigation water through the lake, but the quantities had been at various low rates since 1956, averaging only 5×10^6 m^3 during spring–summer in 1969–1970 when the lake was hypereutrophic. The amount of water needed to improve lake water quality was substantial, but in-lake experiments showed that algal biomass could be reduced in proportion to CRW added (Welch, Buckley, & Bush, 1972). Thus, reduction of lake TP to the mesotrophic boundary of 30 μg/L from around 150 μg/L would require replacing most of the lake water with CRW because phosphorus would still be entering the lake from ground water as well as recycled from bottom sediments.

Funds became available as a Clean Lakes Project from the US Environmental Protection Agency (USEPA), through the Washington Dept. of Ecology (WADOE), in cooperation with the US Bureau of Reclamation (USBR), to increase the transport of CRW through the lake with the goal of reducing spring–summer average TP to 50 μg/L and increasing transparency to 1.5 m. As a result, TP in lower Parker Horn and the South Lake was reduced by one half during 1977–1986 when CRW input averaged 20-fold over the preproject rate, and below 50 μg/L

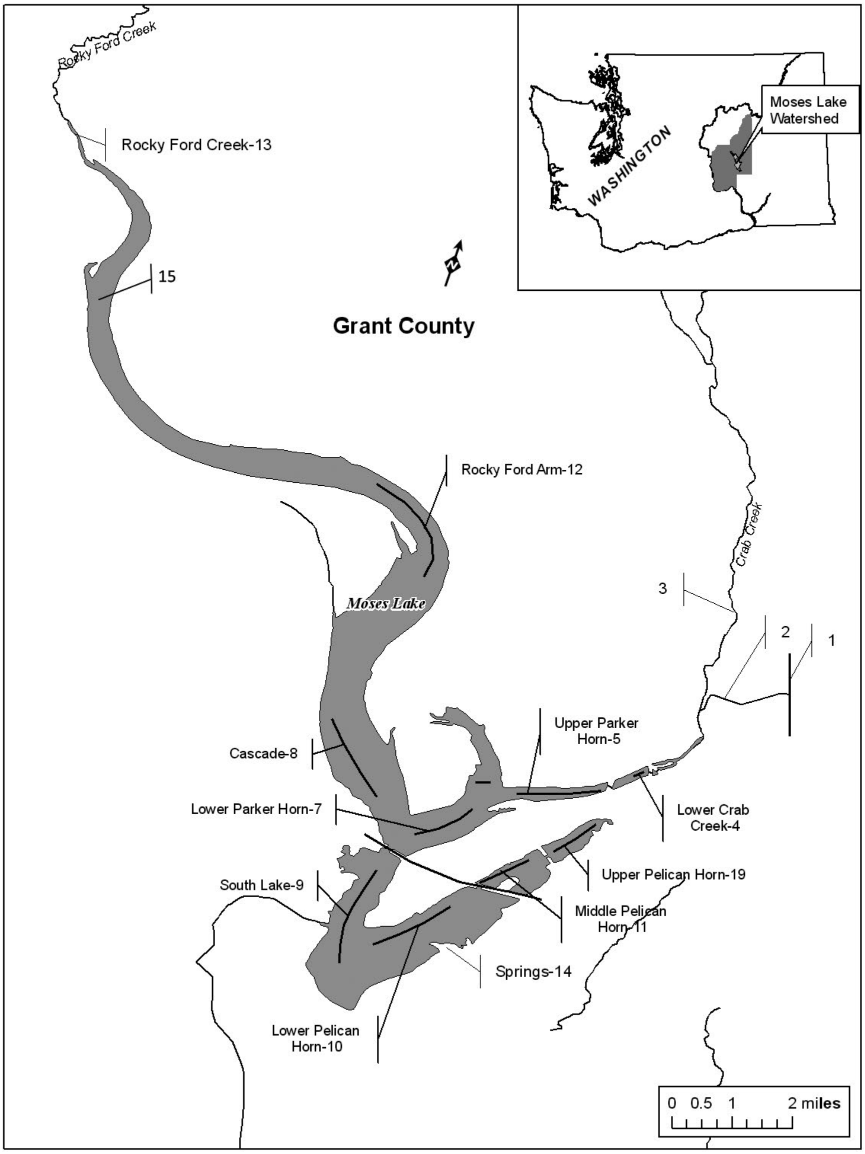

Figure 1. Sampling sites during 2017 and 2018, most similar to those during 1969–1970 and 1977–1988 (Welch et al., 1989). Site 1 is East Low Canal and site 2 is Rocky Coulee Wasteway, routs for CRW inflow. Figure 1 adapted from Welch, 2009.

during 1986–1988 when CRW was over 30-fold above the preproject rate (Welch et al., 1992; Welch, Jones, & Barbiero, 1989). Total P has declined further since 2000 as CRW input increased 60-fold on average over the preproject rate. Effects from the increased inputs since 2000 are emphasized here, as well as the lake's more recent response to lower CRW inputs.

Site Description

Moses Lake has a dendritic shape and is located in eastern Washington. The lake was naturally created by windblown sand dunes that historically dammed CC. The lake level was stabilized with two dams constructed in 1929 and 1963 and is held constant by USBR during April to October at 319 m elevation. The lake has an area of 2,790 ha with a maximum depth of 11.5 m and a mean depth of 5.6 m (Figure 1). Most of the lake (80% of the area) is polymictic, being too shallow to permanently stratify.

There are two tributaries; CC that drains 80% of the watershed (5,265 km^2), which is mostly dry-land agriculture and rangeland, as well as some irrigated land (112 km^2), and Rocky Ford Creek (RFC) that is largely spring-fed. The two streams contribute a similar quantity of inflow to the lake (Jones & Welch, 1990). Under normal inflows, with no CRW added, the lake would have a water residence time of about 1 yr using April–September inflow (Welch et al., 1992).

Parker Horn (sites 5 and 7, Figure 1) and South Lake (site 9 in Figure 1), between Interstate 90 (I-90) and the outlet, represent 27% of the lake's area and data from sites 7 and 9 were used to depict changes in water quality because most of the CRW inflow passes through those sections of the lake (Figure 1). The lower part of the Rocky Ford Arm (RFA) designated as Cascade (site 8 in Figure 1) represents another 14% of the lake's area. Parker Horn (sites 5, 7), Cascade (sites 8), South Lake (sites 9), and lower Pelican Horn (sites 10) represent about half the lake's volume that is most affected by CRW input and is used for recreation. While CRW reached well up into RFA (sites 12 and 15), the effect of dilution was reduced because water transported from lower Parker Horn and South Lake had higher TP than CRW.

Middle and upper Pelican Horn (sites 11 and 19 in Figure 1) are shallow, quite turbid, and largely isolated from most CRW input, although a mixture of CRW and upper Parker Horn water was pumped at 1.42 m^3/s to upper Pelican Horn during portions of the spring–summer period. Analysis showed that CRW had no effect on TP during 1986–1988 or in 2017.

Water Sample Collection and Analysis

The long-term record of CRW inflow, water quality, and trophic state was possible using data from WADOE in 2001, USBR from the 1990s through 2017, and the Moses Lake Irrigation and Rehabilitation District (MLIRD) in 2017 and 2018, as well as data from the University of Washington (UW) during 1969–1970 and 1977–1988. USBR data were from the lake's inlets and outlet, while WADOE and MLIRD data were from multiple sites in the lake, similar to those during 1969–1970 and 1977–1988, as well as inlets, sampled by UW personnel (Welch et al., 1992, 1989).

The lake was sampled twice monthly at eight sites from April through September during 1969–1970 and 1977–1988 (Bush, Welch, & Buchanan, 1972; Welch et al., 1992). Water was collected at a depth of 0.5 m across a series of transects at sites 5, 7, 8, 9, 10, 11, 12, and 19 (Figure 1). Discrete samples were also collected monthly at 0.5 m at similar sites during April–August in 2001 by the WADOE (Carroll, 2006). On the same occasions, samples were collected from RFC (site 13), the inflow to RFA, and from Lower CC (site 4), the inflow to Parker Horn that combined Upper CC (site 3) flow with CRW entering from the ELC (site 1) via RCW (site 2).

Water samples for soluble nutrients were filtered (0.45-μm) at the site during the 1970s–1980s. Soluble reactive phosphorus (SRP) was determined by the acid molybdate heteropoly blue method, NO_3-N by cadmium reduction and total nitrogen (TN) as NO_3-N after persulfate digestion (USEPA, 1979). Total P was determined as SRP in previously frozen samples after persulfate digestion (Strickland & Parsons, 1972). Chlorophyll a (chl) was determined by the fluorometric method, corrected for phaeophytin, through 1986 and spectrophotometrically thereafter. Due to instrument problems, values for 1984 were determined from a regression of algal biovolume on chl using past data. Methods for chl and nutrients during the 1970s–1980s are described in more detail in Welch et al. (1992).

Water samples were collected with a Van Dorn bottle in 2017 and 2018 by MLIRD personnel at a depth of 0.5 m at seven of the eight lake sites (except site 8) during May–September (Figure 1). Sampling at the upper end of RFA (site 15, Figure 1) did not start until July in 2017 but occurred for the full spring–summer in 2018. Inflows were sampled at two sites on CC (sites 1, 4) and ELC (site 1). Lake and inflow sites were sampled usually twice monthly during May through September. Samples were shipped on ice to IEH Analytical

Table 1. Average nutrient concentrations in inflows during spring–summer in 1986–1988 (Welch et al., 1992) compared with 2003–2018

	TP (μG/L)		SRP (μG/L)		NO_3-N (μG/L)	
INFLOWS	1986–1988	2003–2018	1986–1988	2003–2018	1986–1988	2003–2018
Crab Creek	47	48	7	22	685	956
Rocky Ford Creek	164	164	107	129	669	1,403
East Low Canal (CRW)	19	7[b]	4	NA[a]	30	12

Notes. TP, total phosphorus; SRP, soluble reactive phosphorus.

[a]Below detection limit.

[b]2017-2018 only.

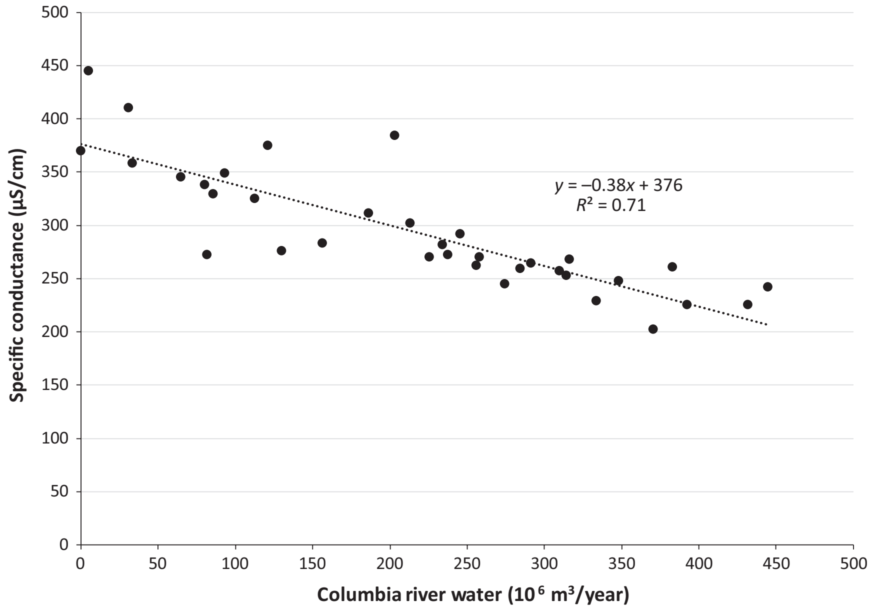

Figure 2. Relation between CRW input and spring–summer average specific conductance in µS/cm in Lower Parker Horn (7) and the South Lake (9) during 1969–1988 from Welch et al. (1992) and South Lake near north outlet during 1995–2018 from USBR. Correlation highly significant at the 0.0001 level.

Laboratories, Seattle, WA, for analysis according to standard methods; 4500PF for TP, 4500NO3F for NO_3-N, and 10200H for Chlorophyll a (chl) following filtration in the IEH laboratory (Eaton, Clesceri, Rice, Greenberg, & Franson, 2005). Specific conductance (SC) was determined in situ with a sonde at all sites coincident with water sampling.

Data in the 1970s and 1980s from Lower Parker Horn (site 7) and South Lake (site 9) are presented as averages of the twice-monthly transect samples at 0.5 m during May–September and were used to indicate lake quality. Transects at 0.5 m integrated the often-patchy distribution of algae and avoided surface scums. Those areas were most affected by CRW that began in April, although CRW reached into Cascade (site 8) and RFA (site 12). Lower CC (site 4) during April–September represented CRW inflow to Parker Horn mixed with normal flow containing background levels of TP in upper CC (site 3). Data for nitrogen and phosphorus from USBR between the mid-1990s and 2018 were from four samples collected near the outlet at South Lake during April–September. The four TP concentrations from USBR (site 9) and ten from MLIRD (sites 7 and 9) in 2017, averaged 25 ± 7 and 27 ± 11 µg/L, respectively, which tended to validate using the 15 years of USBR data as representative of Lower Parker Horn and South Lake trophic state. However, the two sources of data in 2018 (sites 7 and 9) were not comparable because the fourth USBR value was inexplicably high—nearly five times the average for that time in the previous 4 years.

Table 2. Average specific conductance in µS/cm used as tracer of Columbia River water (CRW) throughout Moses Lake

CRW	142
Moses Lake (predilution)	445 (1969–1970)
Crab Creek	491
Rocky Ford Creek	371

Specific conductance (SC) was used to trace CRW in the lake and determine % lake water or % CRW according to (Cooke et al., 2005; Welch & Patmont, 1980):

$$100\ \left[(\mathrm{LW}-\mathrm{ELCW})\ /\ (\mathrm{CCW}-\mathrm{ELCW})\right] = \%\mathrm{LW}$$

where LW is the SC in CC at 491 µS/cm if no CRW were added and %CRW = 100 − %LW.

Water samples for algae identification and enumeration in 2017 and 2018 were collected from the same water sampled for other constituents and preserved with Lugol's solution. Samples for algae were collected in mid-July, early September and early October in 2017 and monthly in 2018. Abundance of algal taxa was determined as cells/ml and expressed as biovolume in mm^3/L based on measured cell volumes of individual species observed (Matthews, Pickens, & Lawrence, 2018, 2019).

Phosphorus loading

External loading during May–September was estimated as average surface inflow volumes from respective sources multiplied by their average TP concentrations during the 5 months. Average flow and TP concentrations in CC and RFC in 2017 were 1.5 and 1.71 m^3/s and 42 and 149 µg/L TP, respectively. Average flow and TP in those inflows were 1.9 and 3.42 m^3/s and 43 and 141 µg/L in 2018. Average TP in CRW was 7 µg/L during both years, observed by MLIRD. A base flow of 0.56 m^3/s was used for RCW (Carroll, 2006).

Average volume-weighted (v-w) lake TP concentration was predicted from v-w inflow TP concentration and

Table 3. Spring–summer Columbia River water (CRW) inflows, half-lake volumes (77.8 × 10^6 m^3) replaced and average spring–summer TP and chl concentrations and transparency (*SD*) in South Lake (9) and Lower Parker Horn (7). USBR data from 9 only

AGENCY	YEARS	CRW (10^6 M^3)	0.5 LAKE VOLUMES	TP (MG/L)	CHL (MG/L)	*SD* (M)
UW	1969–1970	5	0.37	152	58	0.7
UW	1977–1984	118	1.3	74	21	1.4
UW	1986–1988	141	1.6	41	17	1.6
WADOE	2001	284	3.7	19	11	NA
USBR	2002–2016	325	4.8	23	6[a]	NA
MLIRD	2017	93	1.2	27	7	1.4
MLIRD	2018	130	1.7	41	18	1.4

Note. [a]Chl estimated from chl:TP = 0.35

May–September water residence time as T (Brett & Benjamin, 2008):

$$\mathrm{TP}_{\mathrm{lake}} = \mathrm{TP}_{\mathrm{inflow}}/1 + T^{0.5}$$

Predicted TP for the 2 years was compared with observed whole lake v-w TP (Jones & Welch, 1990).

Table 4. Average ratios of soluble and total N and P during spring–summer before and after dilution began in 1977 (from Welch, 2009) compared with recent years (data from USBR) and the Redfield ratio of 7.2 by weight

	1969–1970	1977–1988	2003–2018
NO_3-*N*:SRP	1.2	5.2	9.3
TN:TP	7.5	7.3	17.2

Results

Pattern of dilution with CRW

Large volumes of CRW have been directed mainly through only a portion of the lake. During 2002–2016, the average inflow of CRW was 325 × 10^6 m^3 at a rate of 18 m^3/s over an average of 208 days from April through September. That represents nearly four half-lake volumes contained in the most affected lake sections, Cascade (site 8), Parker Horn (sites 5 and 7), South Lake (site 9), and Lower Pelican Horn (site 10). That flow rate of CRW replaced that half-lake volume (88 × 10^6 m^3) at about 1.8% per day. The normal CC and RFC inflow at 3.7 m^3/s replaced that volume at 0.4% per day. Thus, once CRW was added during the spring and early summer, the resulting mixture of CRW and lake water was replaced very slowly given the relatively slow inflow rate of the two creeks—theoretically on the order of 600 days to replace 90% of the half-lake volume. Thus, some fraction of CRW remained in the lake until the following spring. Ground water also entered the lake, although that flow was less than one-tenth surface inflows.

Nutrient contents of the two major surface inflows to the lake, CC and RFC, are shown in Table 1. Average TP

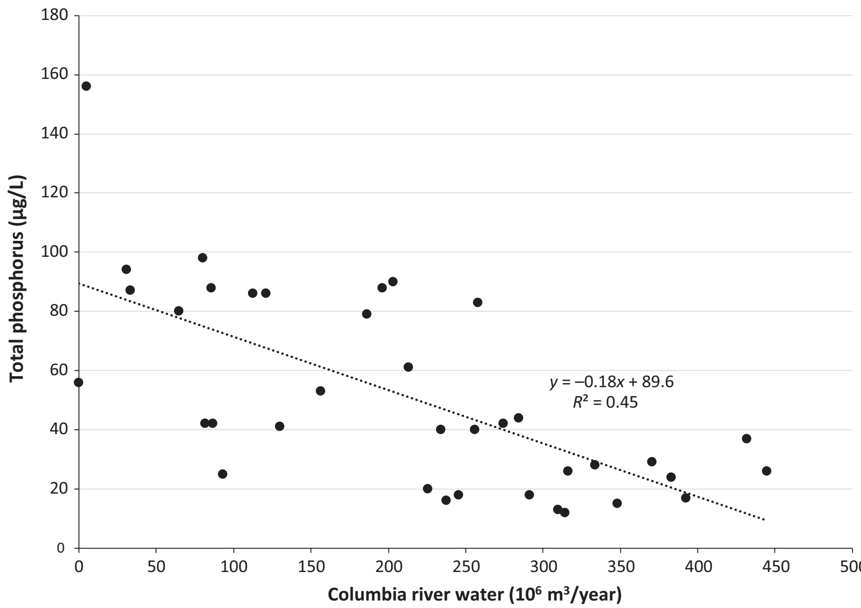

Figure 3. Relation between CRW input and spring–summer average TP concentration in Lower Parker Horn (7) and the South Lake (9) during 1969–1988 from Welch et al. (1992) and South Lake near north outlet during 1995–2018 from USBR. Correlation highly significant at the 0.0001 level.

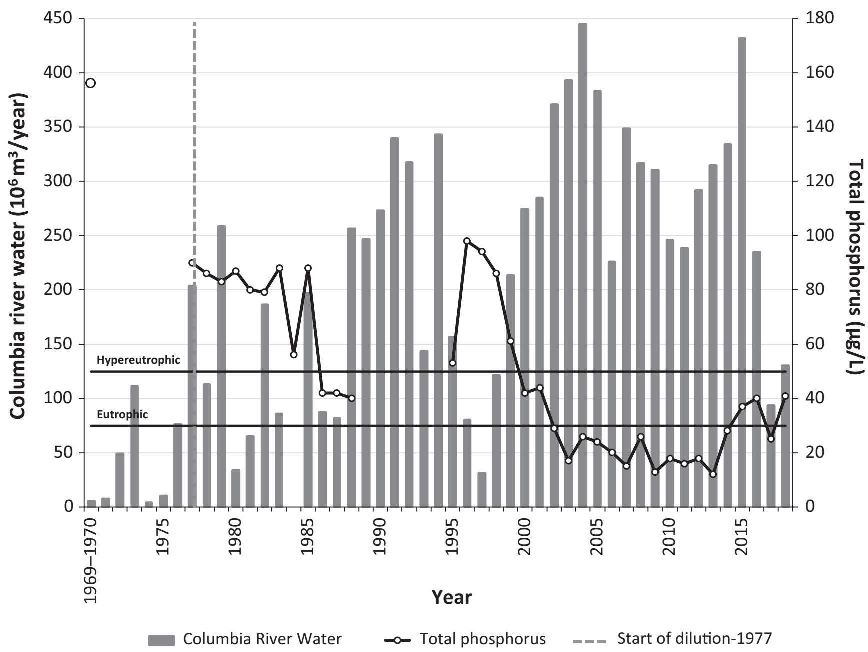

Figure 4. Inflow of Columbia River water into Moses Lake over nearly 50 years and spring–summer average TP concentration in Lower Parker Horn (7) and the South Lake (9) during 1969–1988 (May–September) from Welch et al. (1992) and South Lake near north outlet during mid-April to mid-October, 1995–2018 from USBR.

concentration at the Highway 17 gauging site on CC (site 3) had not changed over the past 15 years from the 1980s, although SRP increased. The CC inflow concentration at site 4 was lowered during the 1970s–1980s when large volumes of CRW, with low TP (19 μg/L) and SRP (4 μg/L), were transported through Parker Horn during the spring–summer periods. Inflow from RFC to upper RFA contained very high TP concentrations with a high fraction that was available SRP (79%) (Table 1). Total P and SRP have remained rather constant in RFC, which is largely spring-fed, but NO_3-N has doubled, raising the NO_3-N:SRP ratio from 6.3 to 10.7 in recent years.

Columbia River water can be traced in the lake because it is low in SC in contrast to CC, which would be the same as the lake SC without dilution (Table 2). Thus, lake SC was highly inversely correlated with CRW input over the past 40 years because CRW inflow with very low SC replaced high SC lake water (Figure 2).

In 2017, SC at Lower Parker (site 7) and South Lake (site 9) averaged 288 μS/cm during May–September. That represents 58% CRW and 42% original lake water, and largely accounts for the low average TP concentration of 27 μg/L observed in that area of the lake. However, TP averaged 41 μg/L during that period in 2018 despite lower average SC at 276 μS/cm and more CRW (62%). There was more internal loading in 2018. While TP was inversely related to CRW inflow over the past 40 years, there were years with relatively high TP due to internal loading, despite substantial CRW (Figure 3).

Despite the relatively low input of CRW (93×10^6 m^3) in 2017, SC in lower Parker Horn (site 7) and South Lake (site 9) increased during the summer from 257 in May to only 297 μS/cm in September, representing a decrease in CRW from 67% to 55%. With slightly more CRW in 2018 (106×10^6 m^3), average SC was lower at those sites in July–September at 273 μS/cm, which represented 63% CRW. That shows how slowly CRW is removed once in the lake. Most of the CRW entered during spring (80% in 2017 and 87% in 2018), which is usually the seasonal pattern. Average SC was even lower at 254 μS/cm or 68% CRW through August in 2016, which had over double the input of CRW (234×10^6 m^3).

Trophic state indicators in 2017–2018 versus 1970s–1980s

The addition of large volumes of CRW has dramatically decreased TP in lower Parker Horn (site 7) and South Lake (site 9) during the 1970s and 1980s. Inputs of CRW increased from 5×10^6 m^3 in 1969–1970, before dilution, to an average of 130×10^6 m^3 during 1977–1988, and TP decreased even further as CRW inputs more than doubled after 2000 to an average of about 300×10^6 m^3 (Table 3, Figure 4).

Algal biomass declined in proportion to lowered TP. Chlorophyll averaged 7 μg/L in lower Parker Horn (site 7) and South Lake (site 9) during spring–summer 2017. Using the ratio of chl:TP (0.35) during 2017, chl during 2002–2016 likely averaged about 6 μg/L (Table 3). The ratio of chl:TP of 0.35 agrees with the average for world lakes at intermediate TP concentrations, indicating that chl is dependent on TP in Moses Lake (Welch & Jacoby, 2004). These chl levels were much lower than in the 1970s–1980s and correspond with lower TP. Total P and chl were much higher in 2018 at 41 and 18 μg/L, respectively, despite slightly more CRW (Table 3).

Transparency increased, but not in proportion to decreased chl. Average spring–summer transparency in lower Parker Horn and South Lake in the past ranged from 0.7 m during 1969–1970, before dilution began, to around 1.5 m during 1977–1988 (Table 3). Observed transparency was consistently 40% greater than predicted from a relationship between chl and transparency (Carlson, 1977). Average observed transparency in lower Parker Horn and South Lake was the same in 2017 and 2018 at 1.4 m, despite average chl at 7 and 18 μg/L, respectively (Table 3). Apparently, transparency is controlled more by non-algal turbidity since dilution began, likely due to resuspension of bottom sediment by carp and wind.

The lake's trophic state changed markedly with dilution. The lake's half volume most affected by CRW is now mesotrophic, as indicated by TP and chl, compared to hypereutrophic before dilution and eutrophic after the start of dilution in the 1970s and 1980s (Table 3). However, transparency, at <2 m, still indicates eutrophy, but some of the turbidity is probably due to nonalgal matter (Nürnberg, 1996).

Trophic state was still hypereutrophic in RFA, which represents 42% of the lake's volume. Total P and chl in upper RFA averaged 86 and 26 μg/L, respectively, in 2017 and 83 and 49 μg/L in 2018. These levels were even higher than averages halfway up RFA (site 12) in the 1970s–1980s at 70 and 22 μg/L, respectively. The chl:TP ratios in upper RFA were around 0.30, about as expected. While CRW reached halfway up RFA (site 12) in the 1970s–1980s, CRW was usually around 30%, much less than in lower Parker and South Lake. With more CRW since 2000, CRW has been much higher in upper RFA—65% during July–September 2018.

Continued addition of low-nutrient CRW has lowered the average summer in-lake ratios of NO_3-N:SRP and TN:TP from 1.2 and 7.5, respectively, in 1969–1970 before dilution, to 5.2 and 7.3 during 1977–1988, to 9.3 and 17.2, during 2003–2018 (Table 4). That increase was due to decreased in-lake phosphorus, as well as increased inflow nitrogen, despite high internal loading—43% of total during 1984–1988 after wastewater diversion (Jones & Welch, 1990). Internal load was a substantial fraction in 2018, but less in 2017. Throughout nearly 50 years of hypereutrophy to mesotrophy, the TN:TP ratio has remained at or above the Redfield ratio of 7.2.

Algae

Cyanobacteria were still abundant in the lake during 2017 and 2018, and averaged 65% of total algal biovolume in lower Parker Horn–South Lake and 77% in RFA during May–September of both years. Maximum cyanobacteria biovolume averaged 11.4 and 55.5 mm^3/L in lower Parker Horn–South Lake and RFA, respectively. Cyanobacteria were more dominant in 2018 at 87% than in 2017 at 43%.

Microcystis aeruginosa (MA) was the major bloom former averaging 78% of maximum cyanobacteria biovolume in lower Parker Horn (site 7), South Lake (site 9), and RFA (sites 12 and 14), despite lower TP since 2000. *Aphanizomenon flos-aquae* (AFA) usually preceded MA seasonally and was more abundant than MA in the 1970s–1980s, before and after dilution began (Welch et al., 1992). It was still dominant in 2005 (Burgoon, 2006). The current dominance of MA may indicate that higher N:P ratios in recent years are favoring that non-N fixing cyanobacterium.

Phosphorus loading

External loading from surface inflows was 8,660 and 13,380 kg in 2017 and 2018, respectively. The higher loading in 2018 was due to a doubling of flow from RFC. Internal loading was higher in 2018 than 2017 as indicated by a greater increase in late summer average TP in lower Parker Horn and South Lake (sites 7 and 9) compared to average inflow TP concentrations from CC; 51 versus 37 μg/L, respectively, in 2018, and 38 versus 30 μg/L in 2017. Thus, lake TP concentration increased above inflow concentration by 14 μg/L in 2018, but was less than inflow in 2017. Similarly, whole lake v-w TP increased from 33 to 46 μg/L in 2017 and 47 to 72 μg/L in 2018. Lake concentrations exceeding inflow concentrations usually indicate internal loading.

Predicted TP in the whole lake from v-w inflow TP concentration and water residence time, also indicated greater internal loading in 2018 than in 2017. Water residences times for the May–September period were 1.07 and 0.75 in 2017 and 2018 from surface inflows of 145 and 208×10^6 m^3, respectively. Resulting v-w inflow TP concentrations for 2017 and 2018 were 60 and 65 μg/L, respectively, giving predicted v-w lake TP concentrations of 30 and 35 μg/L (Brett & Benjamin, 2008). The difference between predicted and observed was more than double in 2018 (35 vs. 72 μg/L) than in 2017 (30 vs. 46 μg/L), with the greater difference in 2018 indicating more internal loading.

Discussion

The addition of high volumes of low-nutrient CRW to central Moses Lake over the past 42 years has consistently maintained a mesotrophic state, especially with the much higher volumes over the past 18 years. Without CRW input, TP concentration in the Parker Horn and South Lake area in 2017 would probably have been more than triple the observed, on the order of 90 μg/L instead of 25 μg/L, judging from TP at South Lake when CRW inputs were very low in 1996–1998 (Figure 4). Even with substantial inputs of CRW, lake TP could be relatively high due usually to high internal loading, for example, increased TP during 2014–2016 with CRW inputs near or above 100×10^6 m^3 (Figure 4).

Inputs of CRW have varied greatly over the past 42 years since the dilution program began, due largely to weather. Average input since 1977 was 226×10^6 m^3 ± 52%. Much of the reason for that variation is indicated by inflow from streams, especially CC. For example, the low average input of CRW in 1996–1998 was 24% of that during 2002–2016, while annual mean CC inflow in 2002–1016 was 29% of that in 1996–1998. Thus, more CRW was routed through the lake in dry years with low CC runoff, than in wet years, due largely to irrigation demand and storage capacity in the downstream reservoir. Nevertheless, the relatively long retention time of CRW in the lake, presumably lasting from 1 year to the next, has allowed rather stable spring–summer TP concentrations even if CRW inputs were low, such as in 2017 with 25 µg/L in Parker Horn and South Lake with only 93×10^6 m^3 of CRW. To maintain a mesotrophic state in at least half the lake volume, CRW input should be at least 200×10^6 m^3. Inputs above that showed spring–summer average TP at 30 µg/L or less (Figure 3).

Despite the lowered trophic state, cyanobacteria still dominated the spring–summer algal crop during 1977–1988, although less (65%) than before the dilution project began (98%, Welch, 2009; Welch et al., 1992). That cyanobacteria averaged 8 mm^3/L and 65% of algal biovolume at lower Parker Horn and South Lake during 1977–1988 was not surprising because TP and chl averaged 65 and 15 µg/L, respectively (Welch et al., 1992). The risk for cyanobacteria dominance was observed to increase sharply at TP above 30 µg/L (Downing, Watson, & McCauley, 2001). Cyanobacteria were still dominant in 2017 and 2018, at 43 and 87% of total biovolume, respectively, consistent with TPs of 25 and 41 µg/L.

Positive net internal loading, even after over 30 years of CRW input and after wastewater diversion, represents a continual threat to lake water quality, particularly as the summer progresses. Continual CRW input annually is essential to dilute internal as well as external loading to maintain acceptable lake water quality for recreation and water use and minimize the frequency and abundance of cyanobacteria blooms. However, lake TP can still increase in mid-to-late summer from internal loading and cause HABs as was the case in 2018, despite slightly more CRW (40%) than in 2017. In contrast, TP was quite low (25 µg/L) in Parker Horn and South Lake in 2017, even with the relatively low CRW input (93×10^6 m^3), likely due to less internal loading (Figure 4).

There has been considerable variability in lake TP for any given CRW inflow, although it was consistently low for inputs above 250×10^6 m^3, as indicated in Figure 3. Variability in lake TP was substantial even with CRW inputs between 100 and 250×10^6 m^3 (Figure 3). Much of that was probably due to internal loading, which varied by ±97% during 1977–1988 largely due to variable year-to-year water column mixing (Jones & Welch, 1990). Internal loading accounted for a lake TP of 90 µg/L in 1985 even with a CRW input of 196×10^6 m^3 (Figure 4).

Internal loading was further verified by laboratory experiments with Moses Lake sediments in 1984 that showed phosphorus release rates ten times higher from anoxic than oxic sediments (Okereke, 1987). The calculated gross internal loading from those laboratory release rates, and respective areas of the lake that were oxic and anoxic, was 7,013 kg during May–September. The average net internal loading calculated directly from mass balance during 1984–1988 was 9,346 kg; 43% of total loading (Jones & Welch, 1990).

Higher inflow NO_3-N concentrations in the two main inflow streams in recent years probably had little effect on algal biomass, because the ratios of NO_3-N:SRP in both CC and RFC were above the Redfield ratio of 7:2. Thus, phosphorus was still the key inflow nutrient determining trophic state. While CRW was also low in NO_3-N, as well as TP and SRP, the inflow ratios of TN:TP in 1977–1988 and 2003–2017 were nevertheless at or above the Redfield ratio of 7.2.

In-lake NO_3-N:SRP ratios were usually well below the Redfield ratio in the 1970s-1980s, indicating short-term growth rate limitation by nitrogen. Nitrate usually decreased to below detection during summer and that was still the case in 2017 and 2018. However, the in-lake TN:TP ratios in the 1970s-1980s were maintained at or slightly above the Redfield ratio despite the low in-lake NO_3-N:SRP ratios and substantial inputs of phosphorus internally, from groundwater and undiluted RFC. The added nitrogen to maintain TN:TP at or above the Redfield ratio was likely produced by nitrogen-fixing AFA, which amounted to 90% of total summer TN loading during 1981–1982 (Welch, 2009). *Aphanizomenon flos-aquae* was usually the dominant cyanobacteria during the 1970s-1980s; for example, it was 48% of average total algal biomass during early summer and 94% on July 8, 1986 (Bouchard, 1989; Welch et al., 1992). *Aphanizomenon flos-aquae* was present in 2017 and 2018, but at a very low fraction of total algal biovolume. Dominance by MA and low biovolume of AFA in Moses Lake in 2017 and 2018 may have been due to the much lower TP and higher N:P ratios in the inflow and lake in recent years.

Increased phosphorus input usually favors nitrogen-fixing cyanobacteria (Gophen, Smith, Aminadav, & Threlkeld, 1999; Havens, 1995). Also, there is agreement among most investigators that nitrogen fixers have the capacity to supply the nitrogen needed to meet available phosphorus, if nitrogen input were low or if nitrogen, as well as phosphorus, input were decreased (Beversdorf, Miller, & McMahon, 2013; Patterson, Schindler, Hecky, Findlay, & Rondeau, 2011). Long-term, continuous phosphorus-only loading to lake 227, Ontario, has kept the lake eutrophic, with nitrogen supplied by nitrogen-fixing cyanobacteria (Higgins et al., 2017; Schindler, Carpenter, Chapra, Hecky, & Orihel, 2016). Similarly, nitrogen fixation by *Aphanizomenon* supplied nearly half the nitrogen input to hypereutrophic Clear Lake, California (Horne & Goldman, 1972). Also, mesocosm experiments in an eutrophic lake demonstrated the capacity of nitrogen fixation to supply enough nitrogen to match the growth potential from P (Vrede et al., 2009). Apparently, nitrogen fixers have that capacity even if TN:TP ratios are above the Redfield ratio (Tonno & Nõges, 2003). Non-nitrogen-fixing MA often succeeds AFA during summer, taking advantage of fixed nitrogen (Beversdorf et al., 2013).

Concerns of added CRW producing adverse downstream eutrophication were raised prior to the start of dilution in 1977 and again recently. Lowering the TP concentration in water leaving Moses Lake should produce less, not greater, eutrophication effects in Potholes Reservoir, immediately downstream. For example, predicted equilibrium TP in Potholes from adding 141×10^6 m^3 of CRW to Moses Lake (average during 1986–1988, Table 3) would be 25 µg/L, given the observed inflow TP of 41 µg/L—the 1986–1988 average in water leaving Moses Lake. Complete mixing in Potholes and a calculated May–September water residence time of 0.38 was assumed. If that 141×10^6 m^3 of CRW and its 19 µg/L TP had entered Potholes directly through a separate canal, together with predilution TP of 152 µg/L and normal outflow from Moses Lake, TP in Potholes would be 45 µg/L—nearly double that if Moses Lake were diluted. High, undiluted Moses Lake water enters Potholes and subsequently to downstream users normally. Thus, not diluting Moses Lake with its usual high internal loading would, theoretically at least, actually increase eutrophication in Potholes. Eutrophication is caused by the resulting nutrient concentration, not by mass loading.

While dilution has been an effective method to improve and maintain Moses Lake quality, there is no year-to-year guarantee of an adequate quantity of CRW because sufficient space may not be available in Potholes in wet years and USBR's first obligation is to downstream irrigators. Other options being considered are to add CRW directly to upper RFA to dilute the high TP (and SRP) entering from RFC, or inject alum directly into the creek to lower its inflow TP concentration. Also, inactivating sediment phosphorus in areas with anoxic bottom water should substantially lower internal loading. Anoxic sediment was observed to release phosphorus at ten times the rate from oxic sediment.

There are other possibilities for lowering inputs of TP. A minor fraction of TP entering through RFC and RCW come from fish hatcheries (Carroll, 2006). Also, a portion of the shoreline is unsewered. On the other hand, some phosphorus inputs have been reduced; wastewater was diverted from Pelican Horn in 1984, and a change from rill to spray irrigation in the early 1970s lowered TP in CC, which has averaged 48 μg/L since 1995—less than half previously (Welch et al., 1992).

Conclusions

The 42-year record of low-P CRW input to Moses lake has demonstrated the effectiveness of lowering lake TP and trophic state when CRW volumes were adequate. If not adequate, lake TP was less effectively lowered, even if inflow TP were decreased proportionately and % CRW were substantially and predictably increased due to TP from internal loading. Cyanobacteria still dominated although their % of total biovolume was usually less than before dilution. The recent switch in dominance from AFA to MA may be due to increased soluble and total N:P ratios in the lake and inflow.

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References

Beversdorf, L. J., Miller, T. R., & McMahon, K. D. (2013). The role of nitrogen fixation in cyanobacterial bloom toxicity in a temperate, eutrophic lake. *PLoS ONE*, *8*, 1–11. https://doi.org/10.1371/journal.pone.0056103

Bouchard, D. (1989). *Carbon dioxide: its role in the succession and buoyancy of blue-green algae at the onset of a bloom in Moses Lake*, MS Thesis. Department of Civil and Environmental Engr., University of Washington, Seattle, WA.

Brett, M. T., & Benjamin, M. (2008). A reassessment of lake phosphorus retention and the nutrient loading concept in limnology. *Freshwater Biology*, *53*, 194–211.

Burgoon, P. S. (2006). *2005 summary of water sampling in Moses Lake*. Wenatchee, WA: WQ Engineering.

Bush, R. M., Welch, E. B., & Buchanan, R. J. (1972). Plankton associations and related hypereutrophic factors in a lake. *Water, Air, and Soil Pollution*, *1*, 257–274.

Carlson, R. E. (1977). A trophic state index for lakes. *Limnology and Oceanography*, *22*, 361–368.

Carroll, J. (2006). *Moses Lake phosphorus-response model and recommendations to reduce phosphorus loading*; Pub. No. 06-03-011. Washington Dept. of Ecology, Olympia, WA.

Cooke, G. D., Welch, E. B., Peterson, S. A., & Nichols, S. A. (2005). *Restoration of lakes and reservoirs*, 3rd ed. Boca Raton, FL: CRC Press.

Downing, J. A., Watson, S. B., & McCauley, E. (2001). Predicting cyanobacteria dominance in lakes. *Canadian Journal of Fisheries and Aquatic Science*, *58*, 1905–1908. https://doi.org/10.1139/f01-143

Eaton, A. D., Clesceri, L. S., Rice, E. W., Greenberg, A. E., & Franson, M. A. H. (2005). *Standard methods for the examination of water and wastewater*, 21st ed. Washington, D.C.: American Public Health Association, Water Environment Federation and American Water Works Association.

Gophen, M., Smith, V. H., Aminadav, N., & Threlkeld, S. T. (1999). Nitrogen deficiency, phosphorus sufficiency, and the invasion of Lake Kinneret, Israel, by the N_2-fixing cyanobacterium *Aphanizomenon ovalisporum*. *Aquatic Sciences*, *61*, 293–306. https://doi.org/10.1007/PL00001326

Havens, K. E. (1995). Secondary nitrogen limitation in a subtropical lake impacted by non-point source agricultural pollution. *Environmental Pollution*, *89*, 241–246. https://doi.org/10.1016/0269-7491(94)00076-P

Higgins, S. N., Patrson, M. J., Hecky, R. E., Schindler, D. W., Venkiteswaran, J. J., & Findlay, D. L. (2017). Biological nitrogen fixation prevents the response of a eutrophic lake to reduced loading of nitrogen: Evidence from a 46-year whole-lake experiment. *Ecosystems*, *21*, 1088–1100. https://doi.org/10.1007/s10021-017-0204-2

Horne, A. J., & Goldman, C. R. (1972). Nitrogen fixation in Clear Lake, Calif: I. seasonal variation and the role of heterocysts. *Limnology and Oceanography*, *17*, 678–692.

Jones, C. A., & Welch, E. B. (1990). Internal phosphorus loading related to mixing and dilution in a dentritic, shallow prairie lake. *Journal of the Water Pollution Control Federation*, *62*, 847–852.

Matthews, R. A., Pickens, J., & Lawrence, E. (2018). *Moses Lake algae monitoring project 2017 final report*. Huxley College, Western Washington University.

Matthews, R. A., Pickens, J., & Lawrence, E. (2019). *Moses Lake algae monitoring project 2018 final report*. Huxley College, Western Washington University.

Nürnberg, G. K. (1996). Trophic state of clear and colored, soft-¨ and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. *Lake and Reservoir Management*, *12*, 432–447. https://doi.org/10.1080/07438149609354283

Okereke, V. O. (1987). *Internal phosphorus loading and water quality projections in Moses Lake*, M.S.E. Thesis. Civil and Environmental Engineering, University of Washington, Seattle, WA.

Patterson, M., Schindler, D. W., Hecky, R. E., Findlay, D. L., & Rondeau, K. J. (2011). Comment: Lake 227 shows clearly that controlling inputs of nitrogen will not reduce or prevent eutrophication of lakes. *Limnology and Oceanography*, *56*, 1545–1547.

Schindler, D. W., Carpenter, S. R., Chapra, S. C., Hecky, R. E., & Orihel, D. M. (2016). Reducing phosphorus to curb eutrophication is a success. *Environmental Science & Technology*, *50*, 8923–8929.

Strickland, J. D. H., & Parsons, T. R. (1972). *A practical handbook of seawater analysis*. Bull. Fish. Res. Board Can. No. 167.

Tonno, I., & Nõges, T. (2003). Nitrogen fixation in a large shallow lake: Rates and initiation conditions. *Hydrobiologia*, *490*, 29–30.

USEPA (1979). *Methods for chemical analysis of water and wastes*. EPA-600/4/79-020. Washington, D.C.

Vrede, T., Ballantyne, A., Mille-Lindblom, C., Algesten, G., Gudasz, C., Lindahl, S., & Brunberg, A. K. (2009). Effects of N:P loading ratios on phytoplankton community composition, primary production and N fixation in a eutrophic lake. *Freshwater Biology*, *54*, 331–334.

Welch, E. B. (2009). Should nitrogen be reduced to manage eutrophication if it is growth limiting? Evidence from Moses Lake. *Lake and Reservoir Management*, *25*, 401–409. https://doi.org/10.1080/07438140903323757

Welch, E. B., Barbiero, R. P., Bouchard, D., & Jones, C. A. (1992). Lake trophic state change and constant algal composition following dilution and diversion. *Ecological Engineering*, *1*, 173–197. https://doi.org/10.1016/0925-8574(92)90001-I

Welch, E. B., Buckley, J. A., & Bush, R. M. (1972). Dilution as an algal bloom control. *Journal of the Water Pollution Control Federation*, *44*, 2245–2265.

Welch, E. B., & Jacoby, J. A. (2004). *Pollutant effects if freshwater: Applied limnology*, 3rd ed. London and New York: Spon Press.

Welch, E. B., Jones, C. A., & Barbiero, R. P. (1989). *Moses Lake quality: results of dilution, sewage diversion and BMPs – 1977 through 1988*. Water Res. Ser. Tech. Rep. 118; Dept Civil & Environ. Eng., Univ. of Washington, Seattle, WA, 65 pp.

Welch, E. B., & Patmont, C. R. (1980). Lake restoration by dilution: Moses Lake, Washington. *Water Research*, *14*, 1317–1325. https://doi.org/10.1016/0043-1354(80)90192-X