A dramatic recovery of Lake Spokane water quality following wastewater phosphorus reduction

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Abstract

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Lake Spokane became hypereutrophic due to nutrient input from a municipal wastewater facility. Following a 1977 reduction in wastewater total phosphorus (TP) from about 5 to 0.5 mg/L, lake water quality and trophic state recovered rather quickly, going from hypereutrophy to meso-eutrophy in the first 7 years. After TP reduction, mean summer (Jun–Oct) inflow TP declined from 86 to 25 μg/L during that 7-year period. Mean summer epilimnetic chlorophyll (Chl) declined from 21 to 11 μg/L, and the mean volume-weighted (v-w) hypolimnetic seasonal minimum dissolved oxygen (DO) increased from 1.4 to 4.5 mg/L over that same period. Recent data (2010–2014) demonstrate continued recovery to meso-oligotrophy with the 5-year average minimum hypolimnetic v-w DO at 6.5 mg/L and mean inflow TP and epilimnetic Chl at 15 and 4 μg/L, respectively. The areal hypolimnetic oxygen deficit (AHOD) rate now averages 0.67 ± 0.12 g/m2 per day, which is 84% less than the pre TP-reduction AHOD (median 4.2 g/m2 per day). This recovery in DO indicators may be the clearest case of recovery from severe eutrophication for a reservoir, which usually have higher AHODs than lakes. The recovery confirms the close link among TP inflow concentration, Chl, and DO in reservoirs, despite their relatively large watersheds and inflows that produce high nutrient loadings compared to natural lakes. The results show that reduction of phosphorus recovered the lake to meso-oligotrophy, even though nitrogen was initially limiting as much or more than phosphorus during hypereutrophy, and despite markedly increased inflow nitrogen since 2000.

Key words: oxygen, recovery, reservoir, TP reduction

There are many accounts of natural lakes recovering from long periods of high nutrient loading following wastewater total phosphorus (TP) reduction or diversion (Sas 1989, Cooke et al. 2005, Jeppesen et al. 2005, Schindler 2012). In virtually all of those cases, recovery was ascribed to phosphorus (P) reductions. Recovery rates have varied from the quick and complete response for the classic case of wastewater diversion from Lake Washington, to shallow lakes with high internal loading and slow responses such as Lake Søbygaard (Edmondson 1994, Sondergaard et al. 1999, 2003). There are few, if any, cases of water quality recovery in large, mainstem reservoirs, however, which would be expected to recover faster than natural lakes due to their usually large inflow volumes and shorter water residence times. Reservoirs have not been the subject of as many long-term studies on TP reduction as have natural lakes. Lake Spokane (also known as Long Lake), a large mainstem reservoir in eastern Washington, may be an exception.

Lake Spokane was studied intensively by R.A. Soltero and associates at Eastern Washington University (EWU) for multiple years before (1972–1977) and after (1978–1985) an 85% reduction of TP in wastewater discharged by the City of Spokane (Soltero et al. 1974, Soltero and Nichols 1984). Data collected during that period were comprehensively assessed in a wasteload allocation study for the Washington Department of Ecology (WDOE) that systematically defined the immediate change in TP loading, reservoir trophic state and dissolved oxygen (DO) concentrations, both before and after the reduction of input TP (Patmont 1987). The earlier data described here are based on Patmont’s analyses and assessment.

Recent data, collected 25 years later, were assessed to determine if recovery had continued. These data show even greater improvement in trophic state during the intervening
years. The decrease in areal hypolimnetic oxygen deficit rate (AHOD) and increase in minimum hypolimnetic DO have been especially dramatic. This record of recovery shows convincingly that DO inventories, as well as chlorophyll (Chl), in at least some large, mainstem reservoirs, can dramatically recover following a substantial reduction in input TP. Despite high inflow volumes from large watersheds producing large annual loads of nutrients and organic matter, the algae and DO in such large waterbodies are, in the end, directly responsive to flow-weighted inflow TP concentration, corrected for sedimentation loss, and not mass load per se (Welch and Jacoby 2004, Brett and Benjamin 2008).

Site description
Lake Spokane is a 37.8 km long, narrow (1.1 km maximum width) reservoir in eastern Washington (Fig. 1). Its area is 2079 ha with a mean depth of 13.2 m and maximum depth at the dam of ∼61.5 m. As in most mainstem reservoirs, Lake Spokane has one main inflow that plunges into the metalimnion between 10 and 15 m, as evidenced by summer conductivity profiles. The euphotic zone, usually 0–7 m in the 1970s–1980s, was calculated by a conductivity mass balance to have a flushing time of 40 days (Patmont 1987). Turbulence generated by wind and inflow was thought to cause mixing between the surface layer and plunging inflow into the metalimnion (Patmont 1987).

The whole reservoir residence time averaged 29 days during June–October 1972–1985. Average whole reservoir residence time during 2010–2014 was similar at 25.2 days and ranged from 14.4 to 36.8 days over this period. Residence times are based on an updated reservoir volume determination of 274 × 10^6 m^3.

Methods
Sampling sites
The pre- and post-wastewater TP reduction water sampling was conducted by EWU between 1972 and 1985 at 5 sites, located at approximately river km 54.4, 61, 67.5, 75.5, and 82.5, starting from the river’s confluence with the Columbia River. The number of different depths sampled ranged from 3 at the most up-reservoir site, where water depth was 10 m, to 12 depths at river km 54.5, where maximum reservoir depth was 52 m. The 5 locations span the long reservoir from the shallowest to deepest sections. Sampling frequency was usually weekly to twice monthly, but with fewer samples during winter.

The sampling program during 2010–2014 had 6 sites: LL0 (52.6 km), LL1 (60.5 km), LL2 (67.7 km), LL3 (74.7 km), LL4 (82.8 km), and LL5 (87.2 km). Site LL5 is considered riverine because water is usually moving except during late summer during extreme low water. Sites LL3 and LL4 are considered transition, and although they thermally stratify, DO is not depleted and maximum water depth is only 20 m. The 3 down-reservoir sites (LL0, LL1, and LL2) are clearly in the well-stratified and DO-deficient lacustrine zone. The riverine, transition, and lacustrine zones represent 10, 29, and 61% of reservoir area, respectively (Fig. 1). The lacustrine and transition sites were at similar positions as in the 1970s and 1980s. Historical data are from annual reports authored by R.A. Soltero and associates (Soltero and Nichols 1984) and were comprehensively analyzed previously by Patmont (1987).

The location of the 2 river sampling locations monitored by WDOE, (1) Spokane River at Riverside State Park and (2) Spokane River at Nine Mile Bridge (Fig. 1), are ∼15 km apart. In recent years, WDOE collected samples at the Nine Mile Bridge location in 2000, 2007–2010, and 2013–2014. R.A. Soltero and associates collected samples in earlier years at this location just downstream from Nine Mile Dam. The Spokane River at Riverside State Park station is considered a long-term water quality monitoring station by WDOE, and samples have been collected at that site by WDOE monthly to twice monthly since 1973.

Water quality constituents
DO was determined every 3 m with a YSI meter during 1972–1985 and during 2010–2014 every 1 m for the top 10 m and every 3 m below that with a Hydrolab datasonde. Volume-weighted (v-w) hypolimnetic DO was determined using values below 15 m for both sets of data during June through October. Depth-strata volumes and mean DOs at those depths throughout the reservoir were used to deter-
mine the v-w hypolimnetic concentrations. AHOD rates, and hypolimnetic minimum DOs were based on time series analyses of these v-w DOs, with the rate of depletion determined by the slope of v-w DOs time curve.

The Chl data from 1972 through 1978 and lake TP from 1981 through 1985 are from samples collected at 2 m intervals in the photic zone (surface to 1% light residual), which was usually 0–7 m (Patmont 1987). Photic zone TP concentrations were closely related to inflow TP, essentially a 1:1 relation between photic zone TP and inflow TP during 1981–1985 (Patmont 1987). Flow-weighted inflow concentrations were used here because they were available for all years in the 1970s and 1980s.

The data for Chl and TP in the reservoir during 2010–2011 are from composited samples collected at 3 m intervals in the photic zone, which was usually 0–9 m. Data for Chl and TP during 2012–2014 are from 0.5 and 5 m in the lacustrine zone and 0.5 m at LL3, LL4, and LL5. The data from both sampling intervals were well within the epilimnion and were used to calculate means within that stratum. For comparison to historic inflow TPs, water column v-w TP concentrations from LL5 during 2010–2014 were used to represent recent inflow TP to the reservoir. LL5 is downstream of all major inflow into the reservoir, which includes the Spokane River and Little Spokane River (Fig. 1) and, are therefore representative of inflow TP.

Data from 1972 to 1985 were intensively evaluated for quality assurance by Patmont (1987). Most of the samples for TP were analyzed according to American Public Health Association (APHA) protocols (1985) by EWU, although other laboratories were involved. TP was analyzed using the persulfate digestion/stannous chloride method, while other laboratories used the same digestion but the more sensitive and stable ascorbic acid method. All TP analyses showed acceptable precision (5–15%); values from EWU were within 10%, although statistical analysis showed a significant negative bias between the reported values and known reference values from EPA (1979). The consistent bias averaged $-12 \pm 4\%$ ($n = 16$); therefore, the TP data from EWU were corrected by the average ratio of reference:reported values (1.15:1) because the analytical procedure was consistent throughout the 1972–1985 period. The standard error of the corrected concentrations was only 3%.

Samples for Chl were primarily analyzed by EWU using only a spectronic-20 instrument from 1972–1979 according to protocols in APHA (1985). Precision was acceptable at 10%, but values determined using reference values and data from 1981 to 1985 with that instrument showed a negative bias of $-34 \pm 3\%$ ($n = 8$), largely believed to be due to the wide 20 nm bandwidth. During 1981–1984, a spectronic-20 and a DU-8 (bandwidth 2 nm) were used for 179 samples. Two relationships from that analysis were used to correct the negative bias in data prior to 1985: (1) between reference values and those analyzed with a spectronic-20 for 1972–1979 data (14% of the corrected value), and (2) between reference values and those analyzed by a DU-8 for 1981–1984 data (9.1% of corrected value). Where values for samples using both instruments were available in 1981, 1982, and 1984, those with the DU-8 (corrected) were used preferentially, and in 1985, DU-8 values compared with the reference had no bias. A more detailed description of Patmont’s quality assurance analysis is provided as supplemental information online.

Analyses for Chl and TP during 2010 and 2011 were performed by the WDOE Manchester Laboratory according to standard methods (Eaton et al. 2005). Minimum detection limits were 5 μg/L for TP and 0.1 μg/L for Chl. Analyses for Chl and TP during 2012–2014 were performed by Aquatic Research, Inc., Seattle, WA, according to standard methods (Eaton et al. 2005). TP was determined by method 4500PI. Minimum detection limits were 2 μg/L for TP and 0.1 μg/L for Chl.

TP data from the Spokane River at Nine Mile Bridge and Riverside Park locations are from WDOE. Samples during 2004–2007 were analyzed by Inductively Coupled Plasma (ICP) analysis according to Method 200.8M (EPA 1994) and since 2008 using method 365.1 (EPA 1993) or 4500PI (Eaton et al. 2005). Total nitrogen (TN) data during 1974–1978 are from Patmont (1987). TN and dissolved inorganic nitrogen (DIN; ammonia and nitrate+nitrite) data are from WDOE. TN samples were determined by WDOE by method 4500NB (Eaton et al. 2005). Ammonia samples were analyzed according to method 4500NH3H and nitrate+nitrite samples by method 4500NO3I (Eaton et al. 2005).

Summer means for TP and Chl from 1972 through 1985 and 1981 through 1985, as reported by Patmont (1987), include data from June through October. Summer means for lake TP and Chl from 2010 through 2014 represent data from June through September because that is the interval normally used to define trophic state. The difference was slight between summer means calculated for June–September versus June–October 2010–2014. Summer means reported herein for the reservoir for 2010–2014 are area-weighted for June–September with the exception of inflow TP concentrations, in which summer mean inflow TP was calculated for June–October for direct comparison with historical data.

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Table 1. Summer (Jun–Oct) mean inflow TP and photic zone Chl concentrations during pre- (1972–1977) and post-TP (1978–1985) reduction and summer (Jun–Sep) photic zone mean Chl in 2010–2011 and epilimnetic mean Chl in 2012–2014. Summer (Jun–Oct) mean inflow TP concentrations for 2010–2014 are water column v-w averages from LL5.

| Year | TP (µg/L) | Chl (µg/L) | Year | TP (µg/L) | Chl (µg/L) | Year | TP (µg/L) | Chl (µg/L) |
|------|-----------|------------|------|-----------|------------|------|-----------|------------|
| 1972 | 71        | 18.7       | 1978 | 28        | 15.0       | 2010 | 16.1      | 4.7        |
| 1973 | 143       | 27.8       | 1979 | 30        | 15.2       | 2011 | 12.5      | 2.7        |
| 1974 | 54        | 17.0       | 1981 | 25        | 11.6       | 2012 | 13.2      | 4.3        |
| 1975 | 60        | 18.4       | 1982 | 23        | 9.4        | 2013 | 17.9      | 3.7        |
| 1977 | 100       | 20.4       | 1983 | 22        | 10.2       | 2014 | 11.5      | 4.4        |
| 1977 | 100       | 20.4       | 1984 | 25        | 8.7        | 1985 | 20        | 7.9        |

Results

Trophic state

Inflow TP during summer (Jun–Oct) declined from an average of 86 ± 38 µg/L (mean ± standard deviation, n = 5) during 1972–1977, before TP reduction, to 25 ± 3.5 µg/L (n = 7) during 1978–1985 after TP reduction (Table 1), excluding 1980 when TP inflow increased to 100 µg/L following the Mt. St. Helens eruption (Patmont 1987). A slight, step-decrease was evident between means in 1978–1981 (28 ± 2.5 µg/L) immediately after TP reduction (1980 excluded) and the 1982–1985 mean (23 ± 2.0 µg/L). TP just downstream of Nine Mile Dam (91% of the inflow volume; Patmont 1987) during June through October continued to show a downward trend, reaching an apparent equilibrium by the 1990s (Fig. 2). TP loading downstream of Nine Mile Dam on the Spokane River, the source of wastewater P, was 95% of the total TP load to the reservoir before and 87% after TP reduction (Patmont 1987). The only other surface inflow between Nine Mile dam and the reservoir that contributed to the remaining TP load (5 and 13%) is the Little Spokane River (Fig. 1). Thus, nearly all surface inputs to the reservoir enter via Nine Mile Dam.

In-reservoir TP was determined only during 1981–1985, but those values showed that photic zone (0–7 m) TP was essentially equal to inflow TP during those years (Patmont 1987). Minimal sedimentation loss (11%) was indicated by mass balance due to the short residence times of that layer (40 days), determined by conductivity distribution, as well as the whole lake’s hydraulic residence time (29 days).

Mean inflow TP (represented by LL5 concentrations) had decreased further to 14.2 ± 2.7 µg/L by summers 2010–2014 (Table 1). Estimated mean inflow TP at Nine Mile Bridge for those years, based on Riverside State Park site values, was similar (13.4 ± 4.2 µg/L). These values were a little over half of the inflow mean observed during the post-TP reduction period in the late 1970s and early 1980s and indicate a basis for continued recovery. The whole reservoir area-weighted epilimnetic mean was slightly less at 11.6 ± 1.5 µg/L during summers 2010–2014. Throughout the reservoir, epilimnetic TP averaged 9.9, 13.7, and 15.3 µg/L in the lacustrine (LL0-LL2), transition (LL3, LL4), and riverine LL5) zones, respectively. The down-reservoir trend of decreasing TP is due partly to sedimentation loss.

Chl in the photic zone ranged from 20 to 140 µg/L and was closely related to inflow TP (Patmont 1987). During the sum-
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mers prior to TP reduction, Chl averaged 20.5 ± 4.3 μg/L and declined to 11.1 ± 3 μg/L during the 7 years after TP reduction (Table 1). The mean was significantly less the last 4 post P-reduction years (1982–1985) at 9.1 ± 1.0 μg/L, relative to 13.9 ± 2 μg/L observed during the first 3 post-reduction years (1978–1981; 1980 excluded; P = 0.036), indicating a trend of gradual recovery associated with the step-wise reduction in inflow TP. However, the short-term, post-reduction decrease in Chl was proportionately larger (35 vs. 18%) than for TP.

The whole reservoir, area-weighted mean epilimnetic Chl decreased further to 4.0 ± 0.8 μg/L by summers 2010–2014 (Table 1). Chl averaged 2.9, 4.4, and 4.0, in the riverine, transition, and lacustrine zones, respectively. There was usually an increasing down-reservoir trend in Chl, opposite to that of TP, in 4 of the 5 years. The down-reservoir trends may have been due to more time for biomass development in the lacustrine zone and particulate P settling in this short residence time system.

Both inflow TP and whole reservoir Chl have continued to substantially decrease since the second part of the post P-reduction period (1982–1985), 23–15 μg/L for TP and 9.1–4.0 μg/L for Chl. The reservoir has recovered from a trophic state of hypereutrophy to meso-oligotrophy over a period of 35 years.

Dissolved oxygen

There was a prompt increase in minimum v-w hypolimnetic DO concentration immediately following wastewater TP reduction (Fig. 3). Mean minimum DO increased from 1.4 ± 1.3 mg/L during 1972–1977 before TP reduction to 2.5 ± 0.6 mg/L during 1978–1981 immediately after TP reduction, and increased further to 4.5 ± 0.35 mg/L during 1982–1985 for a total increase of >3-fold. Minimum DO continued to increase over the intervening 25 years to a mean of 6.5 ± 0.8 mg/L (5.9–7.8 mg/L) during 2010–2014. Thus, the hypolimnetic DO minimum has increased by nearly 5-fold since TP reduction, and that increase was strongly related to decreased inflow TP due to wastewater P reduction (Fig. 3).

Flow can also affect minimum DO. The 2 points well above the line at inflow TPs of 54 and 60 μg/L (Fig. 3) are from 1974 and 1975, which had the 2 highest June–October inflows to the reservoir and shortest water residence times (14.3 and 18 days) during 1966–1985. Also, flow was found to explain most of the variance in the TP to minimum DO relation before and after wastewater TP reduction (Patmont 1987). Mean inflows into Lake Spokane during 2010–2014 were also higher (141 ± 54 m^3/s) than during 1972–1985 (126 ± 47 m^3/s), and residence time was accordingly shorter (25 vs. 29 days). An analysis of all minimum DO and flow data (1970s, 1980s, and recent) showed a weak relation between minimum DO and water residence time (r^2 = 0.30, Fig. 4); however, the negative trend between DO and residence time among the recent years (2010–2014) was stronger (r^2 = 0.69). Thus, TP had a more overriding influence on minimum DO than flow over the long-term, but effects of flow have been more important in recent years.

Although, DO still declines to very low levels at depth, resulting in late summer anoxia near the reservoir bottom, the
v-w means of the whole hypolimnion have greatly increased since the 1980s. Moreover, any further increase expected in the hypolimnetic DO in response to further reductions in inflow TP may be slight or undetectable given year-to-year variation in recent minimum DO (±12% in 2010–2014; Fig. 3 and 4), as well as the trend in recent DOs having been more related to residence time (Fig. 4). Thus, the large increase in minimum DO since inflow TP reduction was clearly due to reduced P and not flow. That is also indicated by a comparison of minimum DOs in 1974 and 2011 (1987), to recent values of 0.54–0.85 g/m² per day, similar to recent recovered rate; King County (2003); Issac et al. (1966), UW data (1971–1975) and Welch and Bouchard (2014); Welch et al. (1980).

AHOD rate also decreased sharply from a range of 2.2–6.3 g/m² per day before TP reduction to 1.8–2.6 g/m² per day immediately after TP reduction, as reported by Patmon (1987), to recent values of 0.54–0.85 g/m² per day (mean 0.67 ± 0.12 g/m² per day) during 2010–2014 (Table 2). Comparing medians of the ranges, which were the only values presented by Patmon, before and after TP reduction (4.2 and 2.2 g/m² per day) with the mean of recent values (0.67 g/m² per day; median 0.67) gives a reduction of 48% immediately after and 84% overall. The AHOD in 2000, the only other recent year with adequate data, was 0.75 g/m² per day. Thus, AHOD has been consistently low for nearly the past decade and a half. In 1981, measured sediment oxygen demand was similar throughout the reservoir, averaging 1.08 ± 0.03 g/m² per day (n = 30), which was 40% of the AHOD that represented total demand, represented by water column phytoplankton production plus sediment (Wagstaff and Soltero 1982, Patmon 1987). By 2000, DO had increased to the extent that total AHOD (0.75 g/m² per day) was less than the 1981 sediment demand.

### Discussion

Published evidence for the recovery of reservoirs from eutrophication is limited. The noteworthy recoveries, or lack thereof, are for natural lakes in which P reduction was the main cause for recovery (Cooke et al. 2005, Jeppesen et al. 2005, Schindler 2012). Large mainstem reservoirs should be expected to recover even faster following P load reduction than natural lakes due to their usually shorter water residence times and possibly to a relatively smaller influence from residual internal P loading. That seems to have been the case for Lake Spokane, which initially recovered rather promptly from hypereutrophy to borderline meso-eutrophy; however, further recovery to meso-oligotrophy may have been a slower process. Despite the short residence time of this reservoir, internal sources of P may have been important and probably declined slowly because evidence of some internal loading still exists. Nevertheless, most of the recovery of DO apparently occurred during the 1990s when inflow TP reached a low apparent equilibrium, or at least by 2000 when the AHOD was similar to present rates.

Remarkably improved water quality and trophic state occurred in Lake Spokane following wastewater TP reduction. Chl levels decreased in proportion to the decrease in inflow...
TP following TP reduction (Fig. 5). Although P was limiting algal biomass at least some of the time before TP reduction, as indicated by an average inflow TN:TP weight ratio of 13:1 during 1972–1977, limitation shifted more strongly to P after TP reduction (1978–1985) when the average inflow TN:TP weight ratio increased to 34:1 (Patmont 1987). This shift was supported by algal growth potential (AGP) test results in which N or N+P limited growth an average of 72% of the time before (1974, 1975, and 1977; mean TN:TP 12:1), to P-only limitation 75% of the time in 1978 (TN:TP 26:1) after TP reduction (Patmont 1987). AGP results were from 8 to 60 assays per season. More frequent N limitation probably accounts for the pre TP-reduction low Chl:TP ratio that is out of line with subsequent points (Fig. 5); TP was simply much higher than required to form the observed biomass. The much higher inflow TN:TP ratio (34:1) after TP reduction resulted in a closer alignment with trophic state boundaries, with P strongly restricting biomass (Fig. 5). Inflow TN:TP ratios have increased even further to an average of 65:1 in 2010–2014, indicating that reduction of P alone improved whole-lake water quality and trophic state, although N limitation occurred some of the time, as indicated by N:P ratios and/or AGP tests under hypereutrophic conditions (Welch 2009, Schindler 2012). Moreover, continued low inflow TP has maintained high lake quality despite markedly increased inflow TN and DIN concentrations (Fig. 2). Flow-weighted mean inflow TN during June–October was 950 μg/L before (1972–1977) and 824 μg/L after (1978–1985) wastewater TP reduction but has steadily increased since 2000 to an average of 1135 ± 96 μg/L during 2010–2014. The 13% difference between before and after TN is about the same as year-to-year variation (12%; 1974, 1975, 1977, and 1978).

The continued reduction in TP and Chl to meso-oligotrophy in 2010–2014 may have happened rather gradually since the late 1980s. The response was rather rapid during the first years after TP reduction, but the pattern may have continued more gradually for the next decade or so as internal sources of P declined. Unfortunately, there are no intervening in-reservoir data from which to accurately determine the rate of response during the later phase of the recovery. Inflow TP concentrations apparently reached an equilibrium shortly after the 7-year, post TP-reduction monitoring period because data from the river farther upstream (Spokane River at Riverside State Park), extrapolated to the Nine Mile Bridge sampling location representing 90% of the inflow to the reservoir, show that inflow TP reached near-current levels by 1990 (Fig. 2). Estimated TP concentrations for the Spokane River at Nine Mile Bridge were calculated based on a relationship between the long-term monitoring dataset at the Riverside State Park station and the observed data at the Nine Mile Bridge station (Fig. 2).

An internal sediment source of P apparently existed because blooms of cyanobacteria occurred in the vicinity of LL4 during 1978–1982 (Soltero and Nichols 1984), and blooms still occur at and between LL4 and LL5, where maximum Chl can reach 20 μg/L. The continued late summer appearance of those blooms indicates that internal P loading in the upper reservoir may have been slow to decline. That conclusion is supported by laboratory-determined sediment P release rates in aquatic plant beds averaging 20 and 7 mg/m² per day under anoxic and oxic conditions, respectively (Owens and Cornwell 2009). While internal loading was not assessed as a source of P during 1972–1985, photic zone TP was observed to increase throughout the summer, which often indicates an internal source as inflow volume decreases, and its effect on reservoir TP is less (Patmont 1987). The photic zone TP was similar to inflow TP on average during June–October 1981–1985, however, which suggests an internal source of P that roughly balanced P losses through sedimentation. A TP mass balance showed an average retention of only 11 ± 9% of the external load (Patmont 1987). Although the short water residence time would have lessened TP retention, near-zero TP retention is unlikely.

Internal loading in eutrophic reservoirs often occurs in the riverine and transition zones, as was the case in DeGray Reservoir, Arkansas, and in Tenkiller Reservoir, Oklahoma (Kennedy et al. 1986, Cooke et al. 2011). In Tenkiller, the average riverine zone TP during summer was double the inflow concentration, resulting in a sizable average internal loading rate of 18 mg/m² per day (Cooke et al. 2011). Had internal loading existed in the upper reservoir zones of Lake Spokane, it probably would have declined gradually following inflow TP reduction, as has often been the case in lakes following P reduction (Sas 1989, Cooke et al. 2005, Jeppesen et al. 2005). The late summer blooms observed...
between LL4 and LL5 and the laboratory-determined sediment P release rates in that area suggest that internal loading still exists. Although probably less than pre P-reduction, internal loading now may even represent a larger fraction of total loading because inflow and epilimnetic TP are much lower than post TP-reduction levels.

The magnitude of hypolimnetic DO recovery is of special interest because (1) there are few cases of DO recovery following TP reduction and none in reservoirs, and (2) the magnitude of DO recovery (AHOD decrease and minimum DO increase) was large. In 2 cases of natural lakes, Onondaga and Washington, AHOD was observed to decline by 49 and 34%, respectively. The AHOD rate in Lake Washington had returned to near the pre-wastewater level by the mid-1980s, and the decrease was less than half that in Lake Spokane for a similar decrease in inflow TP (Table 2). In comparison, AHOD in Lake Sammamish has not changed significantly since before wastewater diversion (Welch and Bouchard 2014), but the fraction of TP input reduction to Lake Sammamish was much less than in the case of Lake Washington.

There is often a tendency to consider hypolimnetic DO in reservoirs to be unlinked to P input due to their usually higher hypolimnetic temperature and higher nutrient (and organic matter) loads. Although average AHOD for reservoirs (n = 33) was 40% higher than for natural lakes at any Chl level (n = 35; Walker 1985), a total organic carbon budget for Lake Spokane showed that phytoplankton production was the principal source of organic matter before (53%) and after (57%) TP reduction (Patmont 1987). Thus, the dramatic reduction in AHOD and increase in hypolimnetic DO in Lake Spokane between 1972 and 1977 and 2010 and 2014, which were proportional to inflow TP reduction, all on the order of 80%, demonstrates the dependency of oxygen resources on P in reservoirs. While some of the year-to-year variation in AHOD and minimum DO may also be related to inflow volume and temperature (Patmont 1987), the long-term changes are primarily attributable to changes in inflow TP concentration. With little recent variation in minimum DOs due to TP (variation from line in Fig. 3 in 2010–2014 < ±14%), there was more effect from flow variation (±38%). Similarly, in hypereutrophic Brownlee Reservoir, low DO and prolonged anoxia were overall related directly to TP (87 μg/L summer epilimnetic mean) but also varied interannually with year-to-year differences in flow and climatic conditions (Nürnberg 2002). Also, increased inflow TP concentration, due to runoff from poultry litter added to the watershed of Tenkiller Reservoir, markedly and directly increased the AHOD (Table 2). Thus, inflow TP can be the key driver to DO depletion or increase, whether to natural lakes or reservoirs.

These results clearly show that wastewater P reduction has alone recovered Lake Spokane from hypereutrophy to meso-oligotrophy as inflow TP continues to decline, despite markedly increasing inflow N concentrations. Moreover, DO has dramatically increased to the degree that much more improvement is unlikely even if inflow TP were to decrease further from its current low of 14 μg/L. There has been some recent debate as to whether inflow N should be reduced, in addition to P, to reduce eutrophication in freshwater (Schindler 2012). These results indicate that N reduction in addition to P reduction would not have been cost-effective to manage Lake Spokane water quality.

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Supplementary Material

Supplemental data for this article can be accessed on the publisher’s website.

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