Global phosphorus retention by river damming

Taylor Maavara[a,b,1], Christopher T. Parsons[a,b], Christine Ridenour[a,b], Severin Stojanovic[a,b], Hans H. Dürr[a,b], Helen R. Powley[a,b], and Philippe Van Cappellen[a,b]

[a]Ecohydrology Research Group, Water Institute, University of Waterloo, Waterloo, ON, Canada N2L 3G1; and [b]Department of Earth and Environmental Sciences, University of Waterloo, Waterloo, ON, Canada N2L 3G1

Edited by Andrea Rinaldo, Ecole Polytechnique Federale Lausanne, Lausanne, Switzerland, and accepted by the Editorial Board November 3, 2015 (received for review June 25, 2015)

More than 70,000 large dams have been built worldwide. With growing water stress and demand for energy, this number will continue to increase in the foreseeable future. Damming greatly modifies the ecological functioning of river systems. In particular, dam reservoirs sequester nutrient elements and, hence, reduce downstream transfer of nutrients to floodplains, lakes, wetlands, and coastal marine environments. Here, we quantify the global impact of dams on the riverine fluxes and speciation of the limiting nutrient phosphorus (P), using a mechanistic modeling approach that accounts for the in-reservoir biogeochemical transformations of P. According to the model calculations, the mass of total P (TP) trapped in reservoirs nearly doubled between 1970 and 2000, reaching 42 Gmol y^{-1}, or 12% of the global river TP load in 2000. Because of the current surge in dam building, we project that by 2030, about 17% of the global river TP load will be sequestered in reservoir sediments. The largest projected increases in TP and reactive P (RP) retention by damming will take place in Asia and South America, especially in the Yangtze, Mekong, and Amazon drainage basins. Despite the large P retention capacity of reservoirs, the export of RP from watersheds will continue to grow unless additional measures are taken to curb anthropogenic P emissions.

phosphorus | river damming | biogeochemical cycles | nutrient retention | eutrophication

The systematic damming of rivers began with the onset of the Industrial Revolution and peaked in the period from 1950 to 1980 (1, 2). After slowing down during the 1990s, the pace of dam building has recently risen again sharply (3). As a consequence, the number of hydroelectric dams with generating capacity >1 MW is expected to nearly double over the next two decades (2). The current surge in dam construction will increase the proportion of rivers that are moderately to severely impacted by flow regulation from about 50% at the end of the 20th century to over 90% by 2030 (3). Homogenization of river flow regimes resulting from damming is a growing, worldwide phenomenon and has been invoked as one of the reasons for the decline in freshwater biodiversity (4).

Another major global driver of environmental change of river systems is enrichment by anthropogenic nutrients, in particular phosphorus (P) (5, 6). Fertilizer use, soil erosion, and the discharge of wastewater have more than doubled the global P load to watersheds compared with the inferred natural baseline (7–10). Because P limits or colimits primary productivity of many aquatic ecosystems, increased river fluxes of P have been identified as a main cause of eutrophication of surface water bodies, including lakes and coastal marine environments (6, 11, 12). River damming and P enrichment are interacting anthropogenic forcings, because sediments accumulating in reservoirs trap P and, thus, reduce the downstream transfer of P along the river continuum (13–15). This raises the question to what extent P retention by dams may offset anthropogenic P enrichment of rivers.

The number of published studies from which P retention efficiencies in dam reservoirs can be obtained is small: an extensive literature search only yields useable data for 155 reservoirs (Dataset S1), that is, less than 0.2% of the ~75,000 dam reservoirs larger than 0.1 km^2 (16). The existing data nonetheless clearly show that even a single dam can significantly alter the flow of P along a river. For example, dam-impounded Lake Kariba (Zambezi River), Lake Diefenbaker (South Saskatchewan River), and Lac d'Orient (Seine River) sequester ~87%, 94%, and 71% of their total P inflows, respectively (17–19). For the 1 million km^2 Lake Winnipeg watershed, 28 reservoirs and lakes accumulate over 90% of the total P load (18). The global retention of P by dams, however, remains poorly constrained (20, 21). Previous estimations have simply applied a correction factor to river P loads to represent retention by dams (22–24). This approach does not distinguish between the various chemical forms of P, nor does it account for differences in reservoir hydraulics or provide information about uncertainties on retention estimates.

Here, we follow a mass balance modeling approach developed previously to calculate the global retention of nutrient silicon by dams (25). The mass balance model represents the key biogeochemical processes controlling P cycling in reservoirs (Fig. 1). The model separates total P (TP) into the following pools: total dissolved P (TDP); particulate organic P (POP); exchangeable P (EP); and unreactive particulate P (UPP). UPP consists mostly of crystalline phosphate minerals that are inert on reservoir-relevant timescales (≤100 y); TDP comprises inorganic and organic forms of P, whereas EP includes orthophosphate and organic P molecules sorbed to or coprecipitated with oxides, clay minerals, and organic matter. Reactive P (RP) is defined as the sum of TDP, EP, and POP; RP represents the potentially bioavailable fraction of TP.

Global predictive relationships for the retention of TP and RP in reservoirs are derived from a Monte Carlo analysis of the model, which accounts for parameter variability within expected

Significance

Phosphorus is an essential nutrient for life. Humans have massively altered the global phosphorus cycle by increasing loading to river systems through fertilizer use, soil erosion, and wastewater discharges. River damming interacts with anthropogenic phosphorus enrichment by trapping a fraction of the phosphorus in reservoir sediments. We estimate that in 2000, 12% of the global river phosphorus load was retained in dam reservoirs. This fraction could increase to 17% by 2030, because of the construction of over 3,700 new dams. Although reservoirs represent a huge phosphorus sink, rising anthropogenic phosphorus emissions continue to outpace the addition of new retention capacity by river damming. The resulting growth in riverine phosphorus export likely contributes to the expanding eutrophication of surface waters worldwide.

Author contributions: T.M., C.T.P., and P.V.C. designed research; T.M., C.T.P., C.R., S.S., and H.R.P. performed research; T.M., C.R., H.H.D., and P.V.C. analyzed data; and T.M. and P.V.C. wrote the paper.

The authors declare no conflict of interest.

This article is a PNAS Direct Submission.

Freely available online through the PNAS open access option.

[1]To whom correspondence should be addressed. Email: tmaavara@uwaterloo.ca.

This article contains supporting information online at www.pnas.org/lookup/suppl/doi:10.1073/pnas.1511797112/-/DCSupplemental.

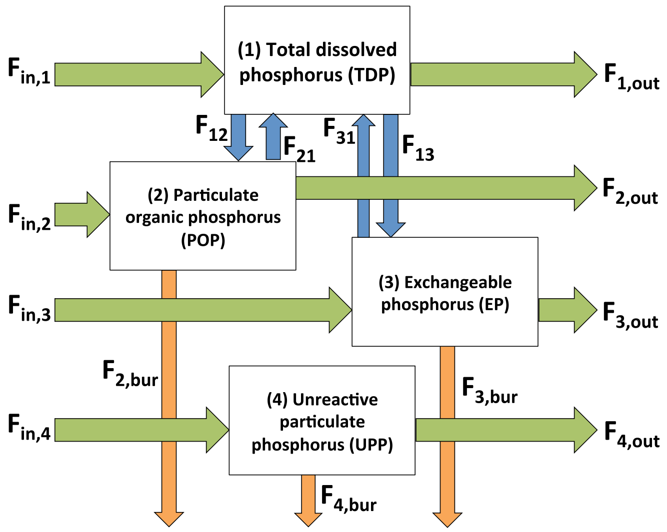

Fig. 1. Mass balance model used to estimate retention of P in reservoirs. $F_{in,i}$ is the influx of the ith P pool into the reservoir, $F_{i,out}$ is the corresponding efflux out of the reservoir, F_{12} represents P fixation by primary productivity, F_{21} represents mineralization of POP, F_{13} and F_{31} are the sorption and desorption rates of dissolved P, and $F_{i,bur}$ is the permanent burial flux of the ith particulate P pool in the reservoir's sediments.

ranges. The relationships are applied to the reservoirs in the Global Reservoirs and Dams (GRanD) database (16), to estimate the sequestration of TP and RP by dams in each of the major river basins of the world. Throughout, P retention efficiencies in a reservoir are defined as

$$R_X = \frac{X_{in} - X_{out}}{X_{in}}, \qquad [1]$$

where R_X is the fractional retention of TP or RP, and X_{in} and X_{out} are the input and output fluxes of TP or RP in units of mass per unit time. Annual amounts of TP and RP retained in a reservoir are then calculated by multiplying the R_X values with the corresponding TP and RP input fluxes from the dam's upstream watershed. The latter are obtained from the Global-NEWS-HD model, which estimates emission yields for dissolved inorganic P (DIP), dissolved organic P (DOP), and particulate P (PP), of which 20% is assumed to be reactive (7, 26). The Global-NEWS-HD yield estimates are based on the biogeophysical characteristics, population density, socioeconomic status, land use, and climatic conditions within the drainage basin (20).

Because the biogeochemical mass balance model explicitly represents the in-reservoir transformations between the different forms of P, it allows us to estimate how dams modify both the total and reactive fluxes of P along rivers. With the proposed approach, we reconstruct global TP and RP retentions by dams in 1970 and 2000 and make projections for 2030. For the latter, we apply the nutrient P loading trends developed for the four Millennium Ecosystem Assessment (MEA) scenarios (27). The results illustrate the evolving role of damming in the continental P cycle and, in particular, the ongoing geographical shift in P retention resulting from the current boom in dam construction.

Results

P Retention in Dam Reservoirs. P retention in lakes and reservoirs correlates with the hydraulic residence time (τ_r) (28–30). Accordingly, τ_r explains more than 45% of the variability of the R_{TP} and R_{RP} values generated by 6,000 Monte Carlo iterations of the P mass balance model. The model-derived R_{TP} and R_{RP} values follow the equation originally proposed by Vollenweider (31) for P retention in natural lakes:

$$R_X = 1 - \frac{1}{1 + \sigma \times \tau_r}, \qquad [2]$$

where σ is a first-order rate constant describing P loss from the water column (see *Supporting Information*, section 4, for a derivation of Eq. **2**). For TP retention in lakes, σ has been related to the relative thickness of the photic zone and the average particle settling velocity (30, 32, 33). Nonlinear least squares regressions yield the following statistically significant average values of σ: 0.801 y^{-1} for R_{TP} ($P < 0.05$) and 0.754 y^{-1} for R_{RP} ($P < 0.05$). The higher σ value for TP reflects the more efficient retention of UPP delivered to reservoirs, compared with the reactive P pools. The resulting difference between R_{TP} and R_{RP} is highest for hydraulic residence times between 0.5 and 1 y.

Preferential accumulation of UPP in reservoirs or, conversely, enhanced relative export of RP from reservoirs, is supported by observations. Salvia-Castellvi et al. (34) found that cascades of small dams in Luxembourg exhibit higher TP retention efficiencies than soluble reactive P, leading to the stepwise increase in TP reactivity after each consecutive dam passage. For 11 out of 16 reservoirs in the Lake Winnipeg drainage basin, Donald et al. (18) similarly found that retention of TP exceeded that of TDP, suggesting that the presence of dams increases the reactive fraction of the riverine P flux.

Global P Retention by Dams: 1970 to 2000. The global, model-predicted retention of TP for 2000 is 42 Gmol y^{-1}, equivalent to 12% of the worldwide river TP load of 349 Gmol y^{-1} (Table 1). The corresponding retention of RP amounts to 18 Gmol y^{-1}. The global annual mass of TP retained in 2000 is almost double that in 1970 (22 Gmol TP y^{-1}), although global TP loading to rivers only increased by 12% over the same time interval. Thus, the growth in TP (and RP) retention during the last three decades of the 20th century primarily reflects the increasing number of dams. The volume of dam reservoirs rose from about 3,000 in 1970 to almost 6,000 km^3 in 2000 (16), whereas the mean reservoir retention efficiencies stayed nearly constant (R_{TP}, ~44%; R_{RP}, ~43%).

During the period from 1970 to 2000, the 3.2 million km^2 drainage basin of the Mississippi River remained the top P-retaining catchment in the world (Fig. 2 *A* and *B*, Table 2, and Dataset S2). In 2000, the 700 reservoirs of the Mississippi River watershed accounted for 5.2% and 5.4% of the global amounts of TP and RP retention by dams, respectively. Other drainage basins with high TP and RP retentions included those of the Zambezi, Nile, Yangtze (Chang Jiang), Volga, and Paraná rivers. The high retentions in the drainage basins of the Zambezi and Volga are explained by a relatively small number of large reservoirs, including the 180-km^3 Lake Kariba along the Zambezi River, and the cascade of reservoirs on the Volga River, including the Volgograd, Rybinsk, and Kuybyshev Reservoirs, each exceeding 25 km^3 in volume.

Projected P Retention by Dams: 2030. Estimates of retention of P in dam reservoirs in 2030 are calculated by combining the four MEA scenarios (27) with the added retention capacity of new hydroelectric dams with generation capacities ≥1 MW that are projected to be completed by 2030 (2). The corresponding global TP retentions fall between 61 and 67 Gmol y^{-1}, or 17% of the global riverine TP loads (Table 1). The RP retentions are in the range 29–36 Gmol y^{-1}. Currently available global projections of river damming do not include smaller reservoirs or reservoirs whose main function is not electricity production. Our projections are therefore likely at the lower end of the potential increase in P retention by 2030.

Table 1. Global retentions of TP and RP by dams, in years 1970, 2000, and 2030

Global estimates	1970	2000	2030AM	2030GO	2030OS	2030TG
Global river TP load, Gmol y^{-1}	312	349	366	384	372	380
Global river RP load, Gmol y^{-1}	113	133	151	175	159	169
TP retained, Gmol y^{-1}	22	42	61	67	62	66
RP retained, Gmol y^{-1}	9	18	29	36	31	35
Fraction of global TP load retained, %	7	12	17	17	17	17
Fraction of global RP load retained, %	8	14	19	21	19	21

The 2030 retentions are calculated by including the new hydraulic dams (>1-MW generating capacity) planned to be completed by 2030 and using the projected 2030 TP and RP river loads for the four MEA scenarios: Adapting Mosaic (AM), Order through Strength (OS), Global Orchestration (GO), and TechnoGarden (TG).

Over the next 15 y, South America, central Africa, and Southeast Asia will experience the greatest growth in P retention by river damming (Fig. 2*C* and Table 2). By 2030, the largest single increase in dam P retention will occur in the Yangtze basin, with up to 2.6 Gmol y^{-1} more RP retained behind 142 new dams. The Yangtze alone will then account for roughly a quarter of the additional mass of RP retained globally. Large increases in TP and RP are also projected for the drainage basins of the Mekong, Salween, and Ganges-Brahmaputra Rivers. In the Mekong River basin, 121 new dams will increase RP retention by 0.7 Gmol y^{-1}. Together, the basins of the Amazon, Paraná, and Tocantins Rivers in South America will retain an additional 0.7 Gmol y^{-1} RP because of the construction of 616 new dams.

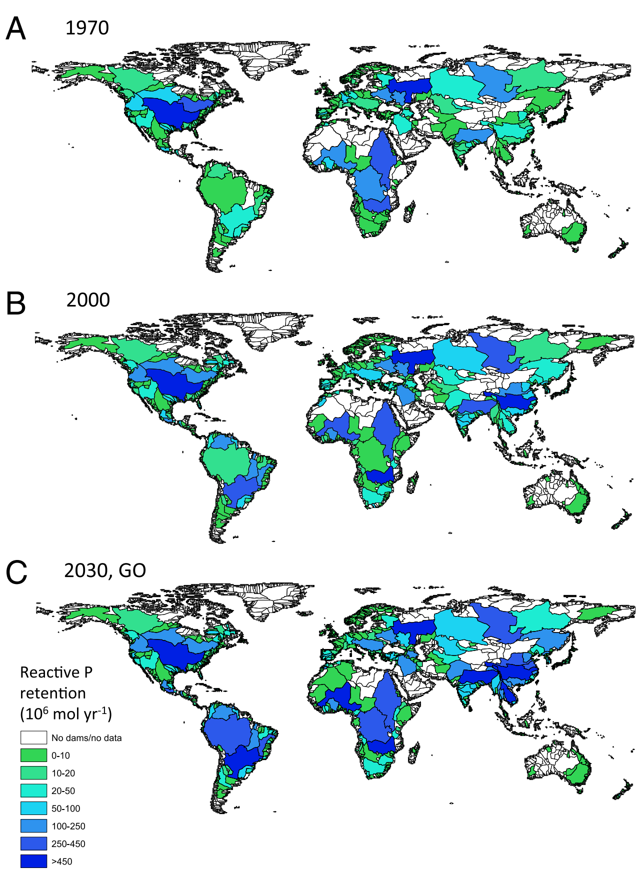

Fig. 2. RP retention by dams in individual watersheds in 1970 (*A*), 2000 (*B*), and 2030 under the GO scenario (*C*). The 2030 RP retentions assume that all dams currently planned or under construction will be completed by 2030 (2). The GO scenario predicts the highest global river P load by 2030 and, hence, yields the largest relative changes in P retention.

In Africa, the Zaire and Zambezi river basins will experience significant increases in P retention attributable to the construction of 30 new dams. The retention of RP by dams in the basin of the Kura River, which empties into the Caspian Sea, should increase by 0.2 Gmol y^{-1} upon completion of 14 new dams.

Discussion

Nutrient enrichment and damming are major anthropogenic pressures on river–floodplain systems and receiving water bodies. By building dams, humans further modify the fluxes and speciation of nutrients along the river continuum (15, 25). In particular, retention in reservoirs can greatly reduce the delivery of P to downstream areas and the coastal zone, influencing regional nutrient limitation patterns, trophic conditions, and food web dynamics (14, 15, 35). For example, the drop in primary production attributable to the near-complete cessation of P supply to the offshore Nile delta region, following the completion of the Aswan High Dam in 1964, is believed to be at the origin of the collapse of the local fishery industry (35). Here, we extend the existing studies on individual reservoirs and watersheds by performing spatially explicit assessments of the global impacts of damming on the riverine P fluxes for the period from 1970 to 2030.

Post-World War II dam construction was particularly intense in North America and Europe, with more than one-third of all dams globally located in the United States by 1970 (16). The geographical hub of dam construction started to shift during the last 30 y of the 20th century. This trend continues to the present day, as new regional economies develop and the need for nonfossil fuel-based energy sources becomes more critical. Current and near-future damming hotspots include western China, the Himalayas and Andes, Brazil, Southeast Asia, and the Balkans, where collectively more than 3,000 major hydropower dams (>1-MW generating capacity) are under construction or planned (2, 3). As a consequence, the global distribution of TP and RP retention by dams in the 21st century will depart significantly from that of the second half of the 20th century.

The global river TP load has at least doubled since prehuman times (7, 8, 26). That is, 50% or more of the average TP flux in rivers is now of anthropogenic origin. In addition, anthropogenic sources deliver relatively more RP to rivers than natural sources (7). From an ecological health perspective, riverine RP is more relevant than TP, because RP represents the P pool that is potentially available for biological assimilation. Although on average individual dam reservoirs retain more than 40% of the inflowing TP and RP, river damming itself had not offset global anthropogenic P enrichment of rivers by the end of the last century. In 2000, the model-predicted worldwide retention of TP by dam reservoirs only represented 12% of the global riverine TP load (Table 1), because (*i*) not all TP entering rivers passes through dam reservoirs and (*ii*) the majority of TP retention currently occurs in smaller reservoirs characterized by relatively short water residence times (≤0.5 y) and, correspondingly, relatively low retention efficiencies (Fig. S1).

Table 2. Top 10 watersheds ranked according to the annual mass of RP retained in their dam reservoirs, for 1970, 2000, and 2030 (GO scenario)

Rank	Watershed	No. of reservoirs	RP load, 10^6 mol/y	RP retained, 10^6 mol/y	Retention, %
1970					
1	Mississippi	546	2,700	1,805	66.9
2	Volga	15	1,423	676	47.5
3	Zambezi	13	734	390	53.1
4	Nile	7	581	376	64.7
5	St. Lawrence	162	1,896	337	17.8
6	Dnepr	5	466	298	63.9
7	Yenisei	3	773	280	36.2
8	Niger	17	602	247	41.0
9	Zaire	6	2,574	221	8.6
10	Ganges-Brahmaputra	43	6,044	152	2.5
2000					
1	Mississippi	700	1,880	920	48.9
2	Zambezi	50	863	531	61.5
3	Volga	17	1,320	500	37.9
4	Yangtze	358	3,758	480	12.8
5	Paraná	70	2,410	357	14.8
6	Ganges-Brahmaputra	83	8,961	322	3.6
7	Yenisei	6	840	267	31.8
8	Niger	52	687	262	38.1
9	Nile	10	624	239	38.3
10	Dnepr	6	438	202	46.1
2030 (GO scenario)					
1	Yangtze	500	8,327	2,898	34.8
2	Mississippi	700	2,294	1,124	49.0
3	Paraná	418	3,912	676	17.3
4	Mekong	140	3,283	650	19.8
5	Zambezi	65	884	649	73.4
6	Ganges-Brahmaputra	483	10,006	621	6.2
7	Niger	74	1,422	568	39.9
8	Volga	17	1,334	506	37.9
9	Zaire	20	2,462	417	16.9
10	Huang He	51	1,033	402	38.9

Nos. of reservoirs, river RP load, and RP retention are provided. An expanded list with the top 150 watersheds for year 2000 can be found in Dataset S2. Only reservoirs listed in the GRanD database and, for 2030, also reservoirs under construction or planned for completion by 2030 are included.

From 1970 to 2000, the fraction of the global river TP load sequestered in reservoirs increased from 7% to 12% (Table 1). This increase is attributed to the construction of about 2,500 new dams during the last three decades of the 20th century, 65% of which have water residence times greater than 6 mo (16) and corresponding average TP retentions in excess of 25% (Fig. S1). By 2030, the retained TP fraction is projected to rise to 17% of the global river TP load, notwithstanding the much higher number of dams (3,782) projected to be built between 2000 and 2030. The more modest increase in post-2000 TP retention per dam, compared with the previous 30-y period, reflects the predominance of hydroelectric dams currently under construction or planned. Hydropower reservoirs generally have shorter water residence times, and correspondingly lower retention efficiencies, than reservoirs of similar size that are primarily used for irrigation or flood control. Of the dams under construction or planned, 63% have reservoirs with water residence times ≤0.1 y (2). In comparison, only 13% of the reservoirs currently included in the GRanD database have water residence times ≤0.1 y.

The combined effects of anthropogenic nutrient enrichment and damming on P export fluxes to the coastal zone are illustrated in Fig. 3 and Fig. S2. The decreases in TP and RP export between 1970 and 2000 observed for Europe are mainly attributable to the ~10% drop in river TP loading following legislation to curb phosphate use in detergents and upgrades to wastewater treatment plants (36, 37). For the same time period, TP export in South America also decreased. However, in this case, damming caused the decline in TP export, as anthropogenic TP loading actually increased. In contrast to TP, RP export in South America increased from 1970 to 2000, because (*i*) RP made up much of the additional anthropogenic P released to rivers and (*ii*) RP tends to be retained less efficiently than UPP in reservoirs. Little change in TP and RP export fluxes are observed for North America, whereas in all other cases, TP and RP exports increased from 1970 to 2000. Thus, at the global scale, the accelerating anthropogenic P release to rivers during the last decades of the 20th century exceeded the added retention capacity of new dams.

Among the four MEA scenarios, the GO and AM yield the largest and lowest riverine TP and RP loads in 2030, respectively (Table 1). Assuming that either (*i*) no new dam construction takes place after 2000 or (*ii*) all dams under construction or planned will be completed by 2030, export fluxes calculated using the GO and AM river loads show that, with the exception of Europe, North America and Australia plus Oceania, the building of new dams in the period from 2000 to 2030 should reduce the export of TP (Fig. 3 and Fig. S2). For Africa and Europe, TP export fluxes are predicted to be lower than in 2000, whereas for Asia, North America, and South America TP, export fluxes under the AM scenario

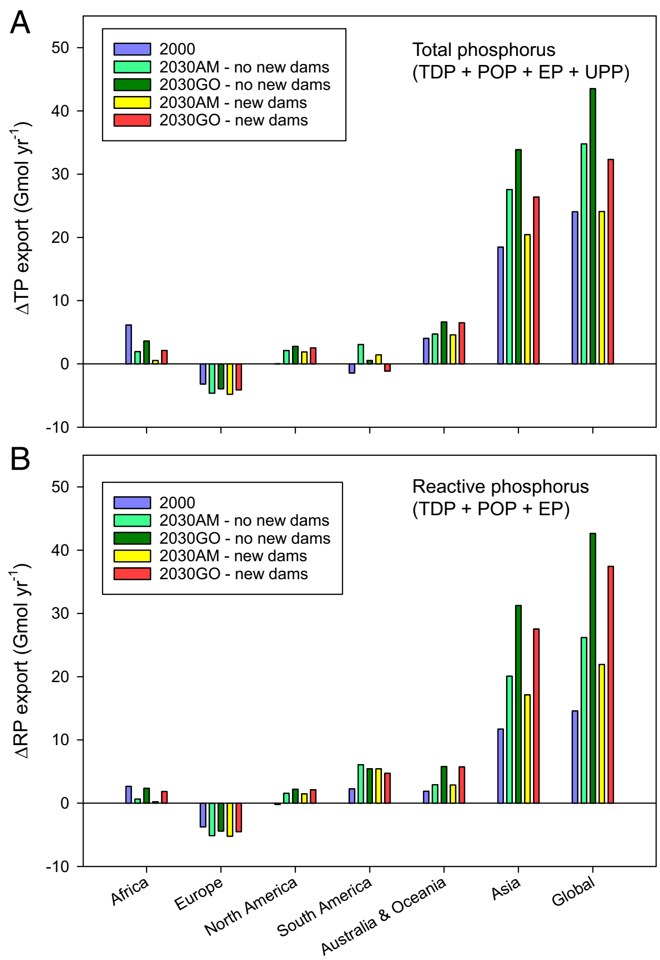

Fig. 3. Changes in riverine export fluxes of TP (*A*) and RP (*B*) to the coastal zone, relative to the corresponding 1970 values. Export fluxes are calculated by subtracting P retained by dam reservoirs in a watershed from the no-dam river P load predicted by the Global-NEWS-HD model. The 2030 scenarios with "no new dams" only account for retention by dams currently in the GRanD database, whereas the 2030 scenarios with "new dams" include the GRanD dams plus those planned to be completed by 2030, as compiled by Zarfl et al. (2).

would remain close to their 2000 values. The current surge in dam construction would therefore appear to be able to largely offset the ongoing and future increases in anthropogenic TP inputs to river systems.

The global export of RP follows a different trajectory, however (Fig. 3*B*). In contrast to TP, even with the construction of new dams, the 2030 export fluxes of RP for South America, Asia, and Australia and Oceania are predicted to substantially exceed the corresponding fluxes in 2000, even under the AM scenario. These upward trends for RP are attributable principally to the fact that anthropogenic sources are mostly delivering reactive P phases to river systems. Hence, for South America, Asia, and Australia and Oceania combined, the RP fraction of the river TP load is estimated to grow from 34% in 2000 to 43% in 2030. Therefore, despite the massive dam-building activity now and in the near future, global anthropogenic RP loading is projected to continue to outpace RP retention until at least 2030. Under the GO scenario, global RP export is expected to be 21% higher in 2030 than in 2000. Such a large global increase in RP export would likely further exacerbate cultural eutrophication of surface water bodies.

Our estimations of TP and RP retention in reservoirs imply that global river damming represents a major anthropogenic perturbation of the continental P cycle. Dams also influence river fluxes of other nutrients, including nitrogen (N) and silicon (Si) (25, 38). Because retention efficiencies by reservoirs differ from one nutrient element to another (18, 25, 39), the presence of dams may modify nutrient stoichiometry along rivers and thereby affect nutrient limitation and food-web dynamics in river-fed aquatic ecosystems (40). The existing evidence suggests dams generally remove P more efficiently than N and Si (18, 39). Damming could therefore be one factor explaining the trend toward more widespread P limitation of coastal waters (41, 42). Given the importance of P as a key, and often limiting, nutrient, as well as the rapid pace of global damming, there is an urgent need to better understand the effects of dams on riverine P fluxes and to fully determine the associated environmental impacts.

Materials and Methods

The biogeochemical processes controlling P cycling in surface water bodies are relatively well understood. The existing knowledge base is thus used to build a P mass balance model that captures the key transformations responsible for changes in P speciation between river inflow and dam outflow (Fig. 1). Note that the model does not account for spatial trends within a reservoir or for subannual variability in P dynamics. The model is not designed to provide a detailed representation of any particular reservoir but rather to perform first-order estimations of annual P sequestration by dams at the river basin scale or higher.

The model assumes that P is supplied to a reservoir via river inflow. For each of the P pools considered in the model (TDP, POP, EP, and UPP), the input is computed as $F_i^{in} = Q \cdot [TP]_{in} \cdot \alpha_i^{in}$, where Q is the volumetric river discharge, $[TP]_{in}$ is the TP concentration of the inflow, and α_i^{in} is the fraction of species i in inflowing TP. The fluxes redistributing P between the TDP, POP, and EP pools, the burial fluxes of the particulate forms of POP, EP, and UPP, and the outflow fluxes of the four pools of P are all assumed to obey first-order kinetics with respect to the corresponding source pool mass. We define burial as the transfer of P below the topmost, active surface layer of sediment, where mineralization and desorption processes remobilize part of the deposited POP and EP. The pools in Fig. 1 are therefore partly located within the water column and partly within the upper, active sediment layer.

The P mass balance model contains 13 adjustable parameters (Table S1). Based on the available literature, probability density functions (PDFs) are assigned to 11 parameters, whereas fixed values are imposed to the remaining two (Table S1). Monte Carlo simulations are carried out by randomly generating 6,000 different parameter combinations from the imposed PDFs. Each individual model run is performed with Runge-Kutta 4 integration and 0.01-y time steps, for the length of time elapsed since dam closure (i.e., if the dam is 20 y old, the model is run for 2,000 time steps). As shown in Fig. S1, the model-predicted R_{TP} and R_{RP} values exhibit positive trends with the hydraulic residence time, τ_r, as expected from the literature (28–30). The trends are fitted to the classic Vollenweider model for P sequestration in lakes (Eq. **2**).

The TP and RP retention relationships (i.e., Eq. **2**) are applied to the dams included in the GRanD database (16). For 1970 and 2000, only dams in operation in those years are considered in the calculations. For the 2030 scenarios, the GRanD database is augmented with the new hydroelectric dams (>1-MW generating capacity) projected to be completed by 2030 (2). Inputs of TP and RP to the reservoirs are obtained by overlaying the year-specific TP and RP watershed yields from the Global-NEWS-HD model (20) onto the GRanD reservoirs in 1970, 2000, and 2030. The yields are then multiplied by the surface areas of the reservoirs' drainage basins to derive the loads in mass per unit time. The 2030 yields used for the individual MEA scenarios are taken from Seitzinger et al. (27). Note that the global estimations of the annual amounts of TP and RP retention by dams at the time points considered include a correction to account for the reservoirs not included in the GRanD database (~24% of total reservoirs in 2000, predominantly those <0.1 km^2). See the *Supporting Information* for full details, including model sensitivity analysis (*Supporting Information*, section 3 and Table S2).

To approximate the uncertainties on global dam P retention estimates associated with the mechanistic modeling approach, we fitted gamma functions to the R_{TP} and R_{RP} distributions produced by the 6,000 Monte Carlo simulations of the model (Fig. S1). A second Monte Carlo analysis was then carried out to calculate global P retentions in which, for each reservoir in GRanD, R_{TP} and R_{RP} values were randomly selected from the corresponding gamma distributions. A total of 20 simulations were carried out, yielding ±7% SDs on the average global TP and RP retentions. Note that these relatively modest uncertainties do not account for the errors associated with the model structure, the Global-NEWS-HD output, and the GRanD database. Model uncertainties on the global TP and RP retentions of the 2030 MEA scenarios are $\pm$20% higher than those of 1970 and 2000, because dams added between 2000 and 2030 tend to have shorter water residence times (95% have $\tau_r \leq 2$ y) than dams built before 2000. The $\pm$20% error estimate is based on the drop in goodness-of-fit of Eq. **2** when only using the results of the Monte Carlo analysis for τ_r between 0 and 2 y.

ACKNOWLEDGMENTS. We thank Christiane Zarfl, Josef Hejzlar, and Jens Hartmann for sharing their databases and Zahra Akbarzadeh for constructive discussions. This project was supported by the Canada Excellence Research Chair (CERC) program and a Natural Sciences and Engineering Research Council of Canada (NSERC) Postgraduate Scholarship (to T.M.).

1. Vörösmarty CJ, et al. (1997) The storage and aging of continental runoff in large reservoir systems of the world. *Ambio* 26(4):210–219.
2. Zarfl C, Lumsdon A, Berlekamp J, Tydecks L, Tockner K (2015) A global boom in hydropower dam construction. *Aquat Sci* 77(1):161–170.
3. Grill G, et al. (2015) An index-based framework for assessing patterns and trends in river fragmentation and flow regulation by global dams at multiple scales. *Environ Res Lett* 10(1):015001.
4. Poff NL, Olden JD, Merritt DM, Pepin DM (2007) Homogenization of regional river dynamics by dams and global biodiversity implications. *Proc Natl Acad Sci USA* 104(14):5732–5737.
5. Smil V (2000) Phosphorus in the environment: Natural flows and human interferences. *Annu Rev Energy Environ* 25:53–88.
6. Correll DL (1998) The role of phosphorus in the eutrophication of receiving waters: A review. *J Environ Qual* 27(2):261–266.
7. Compton J, et al. (2000) Variations in the global phosphorus cycle. *SEPM SP* 66:21–33.
8. Filippelli GM (2002) The global phosphorus cycle. *Rev Mineral Geochem* 48(1): 391–425.
9. Meybeck M (1993) C, N, P and S in rivers: From sources to global inputs. *Interactions of C, N, P and S Biogeochemical Cycles and Global Change*, eds Wollast R, Mackenzie FT, Chou L (Kluwer Academic Publishers, Dordrecht, The Netherlands), pp 163–193.
10. Ruttenberg K (2003) The global phosphorus cycle. *Treatise Geochem* 8:585–643.
11. Schindler DW (1977) Evolution of phosphorus limitation in lakes. *Science* 195(4275): 260–262.
12. Conley DJ, et al. (2009) Controlling eutrophication: Nitrogen and phosphorus. *Science* 323(5917):1014–1015.
13. Harrison JA, Bouwman AF, Mayorga E, Seitzinger S (2010) Magnitudes and sources of dissolved inorganic phosphorus inputs to surface fresh waters and the coastal zone: A new global model. *Global Biogeochem Cycles* 24(1):GB1003.
14. Teodoru C, Wehrli B (2005) Retention of sediments and nutrients in the Iron Gate I Reservoir on the Danube River. *Biogeochemistry* 76(3):539–565.
15. Friedl G, Wüest A (2002) Disrupting biogeochemical cycles - Consequences of damming. *Aquat Sci* 64(1):55–65.
16. Lehner B, et al. (2011) High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. *Front Ecol Environ* 9(9):494–502.
17. Kunz MJ, et al. (2011) Sediment accumulation and carbon, nitrogen, and phosphorus deposition in the large tropical reservoir Lake Kariba (Zambia/Zimbabwe). *J Geophys Res Biogeosci* 116:G03003.
18. Donald DB, Parker BR, Davies J-M, Leavitt PR (2015) Nutrient sequestration in the Lake Winnipeg watershed. *J Great Lakes Res* 41(2):630–642.
19. Garnier J, Leporcq B, Sanchez N, Philippon X (1999) Biogeochemical mass-balances (C, N, P, Si) in three large reservoirs of the Seine Basin (France). *Biogeochemistry* 47(2): 119–146.
20. Mayorga E, et al. (2010) Global Nutrient Export from WaterSheds 2 (NEWS 2): Model development and implementation. *Environ Model Softw* 25(7):837–853.
21. Lerman A, Mackenzie FT, Ver LM (2004) Coupling of the perturbed C-N-P cycles in industrial time. *Aquat Geochem* 10:3–32.
22. Mackenzie FT, Ver LM, Lerman A (2002) Century-scale nitrogen and phosphorus controls of the carbon cycle. *Chem Geol* 190(1-4):13–32.
23. Beusen AHW, Dekkers ALM, Bouwman AF, Ludwig W, Harrison J (2005) Estimation of global river transport of sediments and associated particulate C, N, and P. *Global Biogeochem Cycles* 19(4):GB4S05.
24. Harrison JA, et al. (2005) Dissolved inorganic phosphorus export to the coastal zone: Results from a spatially explicit, global model. *Global Biogeochem Cycles* 19(4): GB4S03.
25. Maavara T, Dürr HH, Van Cappellen P (2014) Worldwide retention of nutrient silicon by river damming: From sparse data set to global estimate. *Global Biogeochem Cycles* 28(8):842–855.
26. Meybeck M (1982) Carbon, nitrogen, and phosphorus transport by world rivers. *Am J Sci* 282(4):401–450.
27. Seitzinger SP, et al. (2010) Global river nutrient export: A scenario analysis of past and future trends. *Global Biogeochem Cycles* 24(4):GB0A08.
28. Kõiv T, Nõges T, Laas A (2011) Phosphorus retention as a function of external loading, hydraulic turnover time, area and relative depth in 54 lakes and reservoirs. *Hydrobiologia* 660(1):105–115.
29. Hejzlar J, Šámalová K, Boers P, Kronvang B (2006) Modelling phosphorus retention in lakes and reservoirs. *Water Air Soil Pollut Focus* 6(5-6):487–494.
30. Brett MT, Benjamin MM (2008) A review and reassessment of lake phosphorus retention and the nutrient loading concept. *Freshw Biol* 53(1):194–211.
31. Vollenweider RA (1975) Input-output models with special reference to the phosphorus loading concept in limnology. *Schweiz Z Hydrol* 37(1):53–84.
32. Müller B, Bryant LD, Matzinger A, Wüest A (2012) Hypolimnetic oxygen depletion in eutrophic lakes. *Environ Sci Technol* 46(18):9964–9971.
33. Müller B, Gächter R, Wüest A (2014) Accelerated water quality improvement during oligotrophication in peri-alpine lakes. *Environ Sci Technol* 48(12):6671–6677.
34. Salvia-Castellvi M, Dohet A, Vander Borght P, Hoffmann L (2001) Control of the eutrophication of the reservoir of Esch-sur-Sûre (Luxembourg): Evaluation of the phosphorus removal by predams. *Hydrobiologia* 459(1-3):61–71.
35. Nixon SW (2003) Replacing the Nile: Are anthropogenic nutrients providing the fertility once brought to the Mediterranean by a great river? *Ambio* 32(1):30–39.
36. Ludwig W, Dumont E, Meybeck M, Heussner S (2009) River discharges of water and nutrients to the Mediterranean and Black Sea: Major drivers for ecosystem changes during past and future decades? *Prog Oceanogr* 80(3):199–217.
37. Van Drecht G, Bouwman AF, Harrison J, Knoop J (2009) Global nitrogen and phosphate in urban wastewater for the period 1970 to 2050. *Global Biogeochem Cycles* 23(4):GB0A03.
38. Harrison JA, et al. (2009) The regional and global significance of nitrogen removal in lakes and reservoirs. *Biogeochemistry* 93(1-2):143–157.
39. Maavara T, et al. (June 4, 2015) Reactive silicon dynamics in a large prairie reservoir (Lake Diefenbaker, Saskatchewan). *J Great Lakes Res*, 10.1016/j.jglr.2015.04.003.
40. Garnier J, Beusen A, Thieu V, Billen G, Bouwman L (2010) N:P:Si nutrient export ratios and ecological consequences in coastal seas evaluated by the ICEP approach. *Global Biogeochem Cycles* 24:GB0A05.
41. Howarth RW, Marino R (2006) Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over three decades. *Limnol Oceanogr* 51(1):364–376.
42. Elser JJ, et al. (2007) Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. *Ecol Lett* 10(12): 1135–1142.
43. Imboden D, Gächter R (1978) A dynamic lake model for trophic state prediction. *Ecol Modell* 4(2):77–98.
44. Imboden DM (1974) Phosphorus model of lake eutrophication. *Limnol Oceanogr* 19(2):297–304.
45. Snodgrass WJ, O'Melia CR (1975) Predictive model for phosphorus in lakes. *Environ Sci Technol* 9(10):937–944.
46. Hudson JJ, Taylor WD, Schindler DW (2000) Phosphate concentrations in lakes. *Nature* 406(6791):54–56.
47. Cotner J, Wetzel R (1992) Uptake of dissolved inorganic and organic phosphorus compounds by phytoplankton and bacterioplankton. *Limnol Oceanogr* 37(2): 232–243.
48. Slomp CP, Van Cappellen P (2007) The global marine phosphorus cycle: Sensitivity to oceanic circulation. *Biogeosciences* 4(2):155–171.
49. Malmaeus JM, Blenckner T, Markensten H, Persson I (2006) Lake phosphorus dynamics and climate warming: A mechanistic model approach. *Ecol Modell* 190(1):1–14.
50. Griffin TT, Ferrara RA (1984) A multicomponent model of phosphorus dynamics in reservoirs. *J Am Water Resour Assoc* 20:777–788.
51. Katsev S, et al. (2006) Factors controlling long-term phosphorus efflux from lake sediments: Exploratory reactive-transport modeling. *Chem Geol* 234:127–147.
52. Gorham E, Boyce FM (1989) Influence of lake surface area and depth upon thermal stratification and the depth of the summer thermocline. *J Great Lakes Res* 15(2): 233–245.
53. Berner EK, Berner RA (1995) *Global Environment: Water, Air, and Geochemical Cycles* (Prentice Hall, Upper Saddle River, NJ).
54. Jansen N, et al. (2010) Dissolved silica mobilization in the conterminous USA. *Chem Geol* 270(1-4):90–109.
55. Harrison JA, Frings PJ, Beusen AHW, Conley DJ, McCrackin ML (2012) Global importance, patterns, and controls of dissolved silica retention in lakes and reservoirs. *Global Biogeochem Cycles* 26:GB2037.
56. Larsen D, Mercier H (1976) Phosphorus retention capacity of lakes. *J Fish Res Board Can* 33(8):1742–1750.
57. Krogerus K, Ekholm P (2003) Phosphorus in settling matter and bottom sediments in lakes loaded by agriculture. *Hydrobiologia* 492:15–28.
58. Moosmann L, Gächter R, Müller B, Wüest A (2006) Is phosphorus retention in autochthonous lake sediments controlled by oxygen or phosphorus? *Limnol Oceanogr* 51(1):763–771.
59. James WF, Barko JW (1997) Net and gross sedimentation in relation to the phosphorus budget of Eau Galle Reservoir, Wisconsin. *Hydrobiologia* 345(1):15–20.
60. Duras J, Hejzlar J (2001) The effect of outflow depth on phosphorus retention in a small, hypertrophic temperate reservoir with short hydraulic residence time. *Int Rev Hydrobiol* 86(6):585–601.