EFFECTS OF FLOODING ON NUTRIENT BUDGETS AND ECOSYSTEM SERVICES

A Thesis to the Committee of Graduate Studies in Partial Fulfillment of the Requirements for the Degree of Master of Science in the Faculty of Arts and Science

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Abstract

Effects of flooding on nutrient budgets and ecosystem services

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Increases in flooding due to anthropogenic influences such as climate change and reservoir creation will undoubtedly impact aquatic ecosystems, affecting physical, chemical, and biological processes. We used two approaches to study these impacts: a whole-ecosystem reservoir flooding experiment and a systematic literature review. In the whole-ecosystem experiment, we analyzed the impact of flooding on nutrient release from stored organic matter in an upland forest. We found that flooded organic matter produced N (nitrogen) and P (phosphorus), but that more N was released relative to P, increasing the N:P ratio over time. In the systematic literature review, we linked small (<10 year recurrence interval) and extreme (>100 year recurrence interval) floods to changes in 10 aquatic ecosystem services. Generally, extreme floods negatively impacted aquatic ecosystem service provisioning, while small floods contributed positively. Overall, we found that flood impacts vary depending on ecosystem properties (organic matter content) and flood characteristics (magnitude).

Keywords: ecosystem services, flooding, nutrients, rivers, reservoirs

Preface

The structure of this thesis is in agreement with the format set by the graduate studies office, Trent University, 2018. It has been written in manuscript format, where Chapters 2 and 3 (listed below) are anticipated to be published, and Chapters 1 and 4 contain introductory and concluding information in context to the research. Due to the structure of the manuscript format, repetition of concepts and themes between chapters was unavoidable but kept to a minimum. Plural syntax was used to acknowledge the other contributing authors. The two chapters and their publication status are as follows:

Chapter 2:

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List of Abbreviations

DO – dissolved oxygen

FLUDEX – FLooded Upland Dynamics Experiment

GHG – greenhouse gas

GPP – gross primary productivity

HAB – harmful algal bloom

IISD-ELA – International Institute for Sustainable Development-Experimental Lakes Area

MA – Millennium Ecosystem Assessment

NPP – net primary productivity

PN – particulate nitrogen

PP – particulate phosphorus

TC – total coliform

TDN – total dissolved nitrogen

TDP – total dissolved phosphorus

TN – total nitrogen

TP – total phosphorus

TSS – total suspended solids

1. General Introduction

Global environmental change is altering how and where floods occur. In some areas, climate change is expected to increase the magnitude and frequency of floods (Burn and Whitfield 2015), while increases in extreme precipitation may extend flooding to areas outside of normal floodplains (Westra et al. 2014). Additionally, global increases in hydropower generation will inevitably inundate some new terrestrial areas during new reservoir construction, although, in Canada, many of the suitable areas for large dams have already been developed (Zarfl et al. 2015). All floods are not created equal and can be characterized in many ways such as by magnitude or duration or by a variety of flood generating mechanisms such as snow melt or heavy precipitation. Flood variability makes assessing the effects of flooding difficult, but previous literature has made some general conclusions. Previous studies attribute flooding to increases in nutrients (Hubbard et al. 2011), erosion and sediment deposition (Morche et al. 2007), declines in water quality (Buda and DeWalle 2009), changes in habitat connectivity (Phelps et al. 2015), and greenhouse gas (GHG) production (Vidon et al. 2016). However, more research is necessary to better understand how flooding affects aquatic ecosystems and the people who rely on aquatic ecosystem services, especially as flood regimes change.

Increased nutrient loading is one major effect that flooding has on aquatic ecosystems. Recent studies have shown that flooding mobilizes large amounts of nitrogen (N) and phosphorus (P), substantially increasing N and P loads in aquatic ecosystems (Carpenter et al. 2017; McCullough et al. 2012). However, most of these studies focus on areas subjected to anthropogenic nutrient loads from agricultural and industrial areas. One aspect of flooding that has not received enough attention is that, when inundated, natural areas may be substantial sources for nutrients. Quantifying these nutrient inputs will help us understand the magnitude of nutrient release from organic matter and the relative importance of inputs to aquatic systems from flooding. Systems receiving large nutrient loads can experience changes in biological community composition, increased primary production, and possibly experience undesirable algal blooms (Paerl et al. 2011). Additionally, biogeochemical changes resulting from flooding may be responsible for the deterioration of water quality and impaired drinking water supplies. Therefore, quantifying N and P release from flooded organic matter will help us understand how reservoir creation and flooding in new areas will contribute to aquatic nutrient loading.

Flooding is an integral part of the normal flow regimes in rivers, but more frequent extreme floods and fewer small floods will change the biophysical characteristics of rivers and influence aquatic ecosystem services. Aquatic ecosystems provide a variety of ecosystem services such as soil formation, water regulation, drinking water, and areas for recreation. Changes in the availability of ecosystem services caused by flooding will depend on the ecosystem and its properties (Terrado et al. 2013). Flooding that occurs as part of a normal flow regime, including occasional extreme floods, will likely enhance ecosystem service provisioning, but increases in the frequency of extreme floods may reduce an ecosystem's ability to recover between flood events, negatively impacting ecosystem services. Therefore, an investigation into the effects of small and extreme floods on aquatic ecosystem services is necessary to begin understanding how changes in flooding will influence people.

The effects of flooding are complex and understanding the consequences of flooding requires multiple methods of evaluation. Some useful methods for assessing the ecological and biogeochemical consequences of flooding are meta-analyses, surveys, empirical, and experimental studies. Quantitative analyses measuring the direct impacts of flooding are essential for understanding the processes contributing to common problems associated with flooding such as nutrient loading, while qualitative analyses will help guide flood-related research based on large scale observations. Flood studies happen by chance due to the stochastic nature of flooding and it can be difficult to capture the information necessary to evaluate a given flood effect. We can augment these field studies with experimental and bench studies to improve our understanding of flooding and the dynamics behind observed flood outcomes. Additionally, we can use existing studies to make new conclusions and draw attention to topics that require more attention.

Here, we use two approaches to gain a better understanding of flood effects in two different flooded areas: newly flooded reservoirs and riverine floodplains. We partnered with the IISD-Experimental Lakes Area (IISD-ELA) to obtain data from a whole-ecosystem flooding experiment and used it to quantify nutrient release from newly flooded organic matter. The results of this study are beneficial for addressing the effects of new reservoir creation on N and P release and the potential impacts that these additional nutrients will have downstream. We also partnered with the American Geophysical Union (AGU) Chapman Conference on Extreme Events to investigate the impacts of flooding on aquatic ecosystem services and submitted a paper to a related special issue publication. We performed a systematic literature review to explore the relationship between flood magnitude (small vs. extreme) and aquatic ecosystem service provisioning. This helped identify research areas that should receive more attention in the future. Through these studies we aimed to answer the following questions:

Chapter 2: Nutrient budgets calculated in floodwaters using data collected as part of a whole-ecosystem experimental manipulation

Question 1: What is the magnitude of nutrient release from flooded organic matter?

Prediction 1: We expect that flooded organic matter will release N and P into the water column.

Prediction 2: We expect that N and P release will be dependent on the quantity of organic carbon stored in reservoirs, with sites having higher carbon storage releasing the most N and P.

Question 2: How does nutrient release differ after multiple floods?

Prediction 1: We expect that N and P fluxes from flooded terrestrial organic matter should decrease after each repeated flooding season, with the highest N and P fluxes occurring in the first year of flooding and diminishing fluxes in subsequent years.

Chapter 3: Gains and losses of aquatic ecosystem services from small and extreme flooding

Question 1: Does flooding have an effect on the provisioning of ecosystem services?

Prediction 1: We expect that flooding will affect the provisioning of aquatic ecosystem services.

Question 2: Do small and extreme floods affect ecosystem services differently?

Prediction 1: We expect that extreme floods will cause losses in ecosystem services and small floods will generally cause gains in ecosystem services.

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2. Nutrient budgets calculated in floodwaters using a whole-ecosystem flooding experiment

Introduction

Precipitation and flooding have been linked to nutrient loading in aquatic ecosystems (Kaushal et al. 2014; Carpenter et al. 2018). When terrestrial lands flood, stored organic matter in flooded sediments may be an important substrate for decomposition (Hall and St. Louis, 2004; Matthews et al. 2005) and a nutrient source during flooding (Grimard and Jones, 1982). Additionally, human influence is changing which areas experience flooding by artificially flooding for reservoir creation and changing precipitation patterns through global climate change. Although the influence of climate change on flooding is uncertain, recent studies have expressed concern that increases in flood occurrence and magnitude are possible in some areas (Burn and Whitfield 2015). Additionally, in an effort to keep up with increasing energy demand, 3,700 new hydropower dams are planned globally for the next 10 to 20 years (Zarfl et al 2015). An increase in flooding of typically dry areas will exacerbate the consequences of excess nutrients in aquatic ecosystems. The FLooded Upland Dynamics Experiment (FLUDEX) at the IISD-Experimental Lakes Area (ELA) in northwestern Ontario aimed to determine whether differing amounts of stored organic matter in low, medium, and high carbon sites affected greenhouse gas production (GHG), a by-product of decomposition, and methyl mercury yield. Although this hypothesis was not supported by GHG budgets (Matthews et al. 2005; Venkiteswaran et al. 2013) or mercury budgets (Hall et al. 2005), the potential relationship between nutrient release and organic matter content has not been explored. Here, we calculate TN (total nitrogen) and TP (total phosphorus) budgets for the three experimentally flooded FLUDEX reservoirs with different amounts of stored organic carbon to quantify the contribution of N (nitrogen) and P (phosphorus) released in the floodwaters.

Reservoirs are created for a number of reasons but mainly to store water or after damming a river to generate hydroelectricity. In an effort to keep up with increasing energy demand, 3,700 new hydropower dams are planned globally for the next 10 to 20 years (Zarfl et al. 2015), some of which will inundate new areas for reservoir creation. Newly flooded areas, from reservoir creation will have more accumulated organic matter available for decomposition than areas that have been repeatedly flooded and should therefore have more nutrients available for release (Paradis and Saint-Laurent, 2017). Once a new area is flooded, saturated soils leach nutrients into the overlying water and increased moisture stimulates microbial metabolism and decomposition of organic matter (Paterson et al. 1997; Ostrofsky and Duthie 1980). Nutrient loads contributed by flooding associated with new reservoir creation will likely be more dramatic than natural flooding due to greater inundation time and depth. N and P pulses occur quickly after flooding but are temporary and P pulses last for 1-16 years (Lucotte et al. 1999; Lepisto, 1995; Ostrofsky and Duthie, 1980). After the initial period of nutrient release, reservoirs often act as nutrient sinks because of greater sedimentation and biotic uptake associated with increases in water residence time. Previous literature focuses on internal reservoir nutrient loading (e.g. Pearce et al. 2017) and reductions of nutrient delivery to downstream years after reservoir creation (e.g. Van Cappellen and Maavara 2016). However, nutrient pulses following reservoir creation are rarely studied despite the fact that nutrient rich water leaving reservoirs may impact downstream ecosystems. Therefore, quantifying changes in N and P during initial and repeated flooding of organic matter will be important for assessing the consequences of reservoir creation and flooding on downstream nutrient dynamics.

Anthropogenic nutrient inputs from sources such as agriculture or industry often do not increase N and P at equal rates and may cause aquatic N:P ratios to shift (Finlay et al. 2013).

Generally, anthropogenic nutrient enrichment has contributed more P than N, reducing N:P ratios (Yan et al 2016). However, several recent studies have shown that older reservoirs retain more P relative to N (Cook et al. 2010; Grantz et al. 2014), therefore causing the N:P ratio to increase in reservoir outflows relative to inflows. Therefore, N:P ratios in aquatic ecosystems are changing, with some systems experiencing higher N:P ratios and others having lower N:P ratios. Consequences of changing N:P ratios include shifts in phytoplankton community, elemental cycling and metabolism, and overall food web structure (Cross et al. 2007; Schindler et al. 2008). In impounded systems, reservoir outflows typically comprise a large proportion of the nutrients entering downstream systems and therefore have the potential to substantially alter downstream nutrient ratios.

We use data collected during a whole-ecosystem flooding experiment at the IISD Experimental Lakes Area (ELA) to quantify N and P in floodwaters. These data provided a unique opportunity to make whole-ecosystem mass budget calculations because of the controlled hydrology and frequent chemistry sampling. We calculated total nitrogen (TN) and total phosphorus (TP) budgets to determine how much N and P were produced during flooding of sites with varying amounts of stored organic carbon in sediments and vegetation. Previous studies have indicated that decomposition of organic matter can act as a source of nutrients in flooded systems (Kaushal et al. 2014; Grimard and Jones, 1982). As a result, we hypothesized that the magnitude and duration of fluxes of nutrients in the experimental reservoirs would be correlated with the amount of carbon and associated nutrients stored in each reservoir. We also used measurements of physical parameters taken within the experimental reservoirs to identify potential mechanisms that explain patterns of N and P release into floodwaters.

Methods

Site description

The Flooded Upland Dynamics Experiment (FLUDEX) was initiated in 1999 at the Experimental Lakes Area in northwestern Ontario. Three upland forest sites with varying amounts of stored organic carbon were selected and flooded from May or June through September from 1999 to 2003 (Appendix A, Figure A1, A2, A3). Sites were characterized as either low (30,900 kg C ha^{-1}), medium (34,900 kg C ha^{-1}), or high (45,860 kg C ha^{-1}) carbon sites based on vegetation surveys and soil cores collected prior to flooding (Bodaly et al. 2004; Hall et al. 2005). The high carbon site had the most potentially labile or non-woody vegetation. This site was relatively flat, with moist soil covering about half of the site and a drier treed community covering the other half. This site was dominated by jack pine forest (*Pinus banksiana*) with an understory composed primarily of *Sphagnum* and Labrador tea (*Ledum*). Bedrock occurred at a mean depth of 35 cm below the soil. The medium carbon site was dominated by dry jack pine forest and with bedrock a mean depth of 47 cm below the soil surface. The low carbon site was a ridge-top site dominated by dry jack pine forest, which covered about three quarters of the site. This site had thin soils with bedrock mean depth of 15cm below the soil surface and some exposed bedrock (Bodaly et al. 2004; Matthews et al. 2005). Mean reservoir depths over the course of the experiment were 1.2 m ± 0.05, 0.9 m ± 0.15, and 1.0 m ± 0.11 in the low, medium, and high carbon reservoirs, respectively. Reservoir surface areas were 0.63 ha, 0.50 ha, and 0.74 ha in the low, medium, and high carbon reservoirs, respectively. Catchment runoff areas varied considerably among the three reservoirs (low carbon: 0.09 ha, medium carbon: 0.73 ha, high carbon: 4.78 ha).

Reservoir set up and hydrology

Reservoirs walls were constructed of gravel, plastic, and plywood (for reservoirs with dike heights less than 1 m) and wood, cement, and plastic (dike height greater than 1 m) built at low-lying site contours. A layer of polyethylene sheeting was sandwiched between plywood to prevent seepage through the walls. Reservoirs were flooded annually from about May to September from 1999 to 2003. Water was pumped into each reservoir through aluminum irrigation pipes from nearby oligotrophic Roddy Lake and flows were managed using manually controlled valves (Hall et al. 2005). In-line flow meters were installed in the pipes leading to each reservoir to measure input volumes. Water exited reservoirs over v-notch weirs installed in dike walls. Water level recorders in each site were used to calculate reservoir volume and water volumes leaving over v-notch weirs (Venkiteswaran et al. 2013). Reservoirs were drained at the end of each flooding season, usually the end of September, from valves at the bottom of reservoir walls (Hall et al. 2005). Precipitation data were collected at the ELA meteorological site which is located less than 1 km from reservoirs (Matthews et al. 2005). Evaporation was measured using Class A evaporation pans in reservoirs and at the ELA meteorological site (Venkiteswaran et al. 2013). Evaporation occurring on rain-free days was used to estimate losses from the reservoirs and was approximately 2.2 mm day^{-1}.

Chemistry

Water chemistry was sampled weekly during the first month of each flooding season and then biweekly thereafter at the inflow, weir outflow, and within the reservoirs. Samples were collected at more than ten sites distributed throughout each reservoir at depths from 0.5 to 2 m. Analyses for particulate phosphorus (PP), total dissolved phosphorus (TDP), particulate nitrogen (PN), total dissolved N (TDN), NH_4, NO_2, and NO_3 used methods described in Stainton et al. (1977). Dissolved N and P were operationally defined as passing through a GF/C filter. Total nitrogen (TN) and total phosphorus (TP) were calculated by summing dissolved and particulate fractions of N and P. TP and TN were measured at different depths within each reservoir and separated into surface (≤ 0.5 m), middle (0.6 – 1m), and bottom (1.1 – 2m) depths. A YSI temperature/oxygen probe was used to measure dissolved oxygen and temperature at three depths: 0.5 m (surface), 1.0 m (middle), and 1.5 m (bottom) at various sampling sites within reservoirs (Hall et al. 2005).

Nutrient Budgets

Annual water and nutrient budgets were constructed based on previous calculations for carbon (Matthews et al 2005) and mercury (Hall et al. 2005) where:

$$\text{Net}_{(\text{Water, TN, TP})} = \sum O_{(\text{Water, TN, TP})} - \sum I_{(\text{Water, TN, TP})}$$

where $\sum I_{(\text{Water, TN, TP})}$ was the sum of inputs for water, TN, or TP and $\sum O_{(\text{Water, TN, TP})}$ was the sum of the outflows in each year. Inputs included pumped inflow, precipitation, and runoff. Outputs included weir outflow, seepage through reservoir walls and bedrock cracks, evaporation (water budget only), and drain outflow. Seepage was not measured directly, therefore, it was calculated as the residual term in the water budget. We did not account for nitrogen gas phases in balances because there was no evidence of denitrification and N loss to the atmosphere was considered negligible (Hendzel et al. 2005). Significant nitrogen fixation was also considered unlikely because TN:TP ratios were greater than a mass ratio of 10, which is above levels at which nitrogen fixing bacteria predominate (Flett et al. 1980). Nitrogen fixing cyanobacteria were not observed in any of the reservoirs for the duration of the experiment (D. Findlay, unpublished data).

The main sources of error in TN and TP budgets were estimated water budget components. TN and TP error were approximately 1.4 and 1.7 %, respectively. Weir outflow, water level, and precipitation were measured continuously using standard procedures and equipment. Therefore, errors associated with these components are likely small and have been estimated to be about 5%. Error was estimated at 15% for evaporation pans and 18% for estimated direct runoff from the Lake 239 East Inflow (Hall et al. 2005; Winter, 1981).

Inflows

Pump

Pumped inflow volume was measured daily using inline flow meters. Chemistry sampling was completed at the inflow pipe weekly for the first month of flooding and then biweekly throughout the rest of the flooding season. Mass inputs of N and P were estimated by multiplying inflow volumes by the average TN and TP concentrations between chemistry sampling dates.

Precipitation

Precipitation volumes entering each reservoir were estimated using data collected at the ELA meteorological station, which is less than 1 km from the reservoirs. Precipitation was recorded daily using standard and recording gauges. Rainfall amounts were multiplied by reservoir surface areas to estimate the volume of precipitation entering each reservoir. Analyses of precipitation chemistry sampling occurred on an event-based schedule, when collected precipitation volumes exceeded 850 mL. TN and TP concentrations were multiplied by precipitation volumes to determine nutrient inputs.

Runoff

Direct runoff volume from upland areas in reservoir catchments was calculated using direct runoff areas and precipitation depth. Direct runoff areas for individual reservoirs were delineated using topographical maps based on aerial photographs taken in 1982 and 1991 (Hall et al. 2005). Runoff chemistry was determined using weekly data collected from the nearby Lake 239 East Inflow (EIF). This site had a similar forest composition to the FLUDEX uplands and contained no wetlands (Parker et al. 2009). TN and TP masses were estimated as mass per area at Lake 239 EIF and then multiplied by each reservoir runoff area. This estimation assumes that soil and vegetation uptake were similar in reservoir catchments to the L239 EIF upland areas.

Outflows

Weir

Water volumes leaving reservoirs over the v-notch weirs were measured continuously using standard current meter and volumetric methods. Water samples were taken from weir outflows weekly during the first month of flooding and then biweekly throughout the rest of the flooding season. The volume of water leaving the reservoirs over weirs between chemistry sampling dates was summed and then multiplied by average TN and TP concentrations in the intervening period.

Drain

Reservoirs were drained at the end of each flooding season to protect the dike walls and mimic drawdown in shallow areas of hydroelectric reservoirs during the winter. Once water levels dropped below the base of the v-notched weirs, remaining water was removed using drain pipes at the lowest point of each reservoir. Drain volumes were not measured explicitly but were estimated using the reservoir volume prior to drawdown, calculated using maps and water level loggers (Hall et al. 2005). Chemistry sampling only occurred at the drain in 1999 and 2000, therefore drain TN and TP concentrations were estimated from water volumes and weir concentrations for all years to maintain consistency. TN and TP concentrations at the weir in each reservoir on the last sampling date of each flooding season were used as a proxy for drain concentration. At this point in the flooding season, TN and TP concentrations were similar throughout all depths of each reservoir and the weir concentration was representative of the concentrations found throughout the reservoir at drawdown. The weir concentration in each reservoir on the last sampling date was then multiplied by the volume in each reservoir to estimate TN and TP masses leaving through the drain.

Seepage

Seepage losses of water under or through the reservoir walls were estimated as the residual term in the water balances. Seepage estimates were verified using annual seepage surveys, where seepage was channeled into small streams and periodically measured. Chemistry in the seepage was not measured explicitly but was estimated using weir concentrations. Daily seepage was estimated by subtracting daily reservoir water inputs from daily outputs via the weir and evaporation. Daily seepage volumes were then totaled between chemistry sampling dates and multiplied by weir nutrient concentrations to find TN and TP masses leaving the reservoirs via seepage. In the low carbon reservoir, water was also lost to a fracture in the bedrock (Hall et al. 2005). As a result, seepage in the low carbon reservoir also contained water losses to the bedrock fracture because seepage was estimated daily as the residual term in the water budget.

Periphyton

Dense mats of periphyton developed in the FLUDEX reservoirs, growing on flooded trees and vegetation. Given its potential importance as a store of N and P, we estimated the annual mass of N and P stored in periphyton in each reservoir for the years 2000-2003. Wooden dowels were hung in each reservoir prior to flooding to serve as a substrate for periphyton collection. From 2000 to 2002, dowels from each reservoir were collected up to four times per flooding season. In 2003, dowels were only collected once at the end of the flooding season in September. Periphyton was washed from the surface of the dowels and elemental composition was determined by combining samples from the three dowels from each reservoir. Periphyton samples were homogenized in a blender and analyzed for N and P composition. Total masses of N and P stored in periphyton in each reservoir was estimated by extrapolating N and P mass per area on dowels over the entire submerged tree surface area in each reservoir. Submerged tree surface area was estimated by collecting vegetation from areas of sparse, medium, and densely vegetated areas at multiple distances above the ground and mapping the distribution of these vegetation densities over the area of each reservoir. Submerged tree surface areas calculated per reservoir area were 3.03 m^2 m^{-2}, 3.25 m^2 m^{-2}, and 2.92 m^2 m^{-2} in the low, medium, and high carbon reservoirs, respectively.

Statistical analyses

We used a Friedman's F test to assess whether TN and TP fluxes were different among the low, medium, and high carbon reservoirs. We also used paired t-tests to determine if there were differences in N:P flux ratios among the three reservoirs. Three paired t-tests were performed to assess differences between the low and high carbon reservoirs, the low and medium carbon reservoirs, and the medium and high carbon reservoirs.

Results

Hydrology

In all reservoirs, the greatest proportion of the water input was from the pumped inflow (Table 2.1). In the medium and high carbon reservoirs, the greatest portion of the outputs was the weir outflow. Seepage was the largest proportion of output in the low carbon reservoir due to cracks in the bedrock. In the high carbon reservoir, the pumped inflow made up 91% of inflows and weir outflow made up 70% of outflows, on average. Other components such as seepage through dike walls (20%), evaporation (7%), and drained water (7%) made up much less of the total outputs and runoff (5%) and precipitation (3%) made up small proportions of the inputs. On average, the medium carbon reservoir inputs were comprised of 95% pumped inflow, 0.9% runoff, and 3% precipitation. Average outputs in the medium carbon reservoir were 70% weir outflow, 13% seepage, 2% evaporation, and 6% drained water. In the low carbon reservoir, inputs included 97% pumped inflow, 0.09% runoff, and 3% precipitation, on average. Outflows in the low carbon reservoir were comprised of 38% weir outflow, 0.02% evaporation, 7% drained water, and 53% was lost to seepage and the bedrock fracture.

Concentrations of nutrients in inflows and outflows

TN and TP concentrations in the reservoir outflows always exceeded inflow concentrations, indicating that the reservoirs were a nutrient source throughout the 5 years of flooding (Figures 2.1 and 2.2). Generally, the greatest differences in nutrient concentrations between the inflow and outflow occurred in 1999, the first year of flooding. The highest outflow TN concentration in the low carbon reservoir was 705 μg L^{-1} and occurred within the first week of flooding in 1999 (Figure 2.1). Outflow concentrations in the medium carbon reservoir were greatest during the first week of flooding in 1999 (634 μg L^{-1}) and 2003 (628 μg L^{-1}, Fig 1). TN concentrations in the high carbon reservoir outflow were relatively constant with an average of 427 μg L^{-1} throughout the first four years of flooding but increased to an average of 749 μg L^{-1} in 2003. TP concentrations in outflows were also almost always higher than inflows (Figure 2.2). The highest outflow concentrations were measured in the first flooding season for all reservoirs. The highest TP concentrations were measured in the first four weeks of flooding in 1999, when outflow TP concentrations reached 53 μg L^{-1}, 64 μg L^{-1}, and 39 μg L^{-1} in the low, medium, and high carbon reservoirs, respectively (Figure 2.2) After the first twelve weeks of flooding, TP concentrations decreased and were close to average values for the low, medium, and high carbon reservoirs in 2000-2003 (13 μg L^{-1}, 17 μg L^{-1}, and 18 μg L^{-1}, respectively).

Molar TN:TP varied between 12 and 64 and was generally 2 times larger in the inflow than the outflow in all years (Figure 2.3). In all reservoirs, outflow TN:TP was lowest in the beginning of each flooding season and increased toward the end. In the low and medium carbon reservoirs, TN:TP in the outflow was lowest on average in the first flooding season in 1999 (21 and 19, respectively) and increased with each flooding season. In the high carbon reservoir, TN:TP in the outflow was also lowest on average in 1999 (19), but slightly increased to about 23 in 2000 and remained steady in 2001 and 2002. In 2003, the high carbon outflow reached an average of 34.

Internal cycling/conditions

Average water temperatures in the reservoirs were within 1.5ºC of each other and there was no thermal stratification (Figure 2.4). Average annual water temperature in all reservoirs was 18.2 ± 3.2 in 1999, 18.7 ± 3 in 2000, 20.0 ± 2.0 in 2001, 20.5 ± 2.3 in 2002, and 20.3 ± 2.8 in 2003. Generally, temperatures were highest in the middle of each flooding season or about 4 weeks after flooding began (Figure 2.4). DO concentrations were lowest in all reservoirs during the first flooding season. Especially low DO occurred during the first four weeks of flooding and bottom DO concentrations reached close to 0 mg L^{-1} (Figure 2.5). This trend was not found in subsequent years. DO concentrations gradually increased in all reservoirs in each subsequent flooding season. Average DO concentration increased from 3.83 mg L^{-1} in 1999 to 6.79 mg L^{-1} in 2003 in the low carbon reservoir, 2.23 to 5.74 mg L^{-1} in the medium carbon reservoir, 2.97 to 5.88 mg L^{-1} in the high carbon reservoir. The DO concentrations within the low carbon reservoir became similar at all depths in 2003 where DO concentrations were generally the highest (Figure 2.5). Linear regression analysis showed that annual mean DO and TP concentrations were related in all reservoirs (low C: R^2= 0.97, p=0.002; med C: R^2 = 0.90, p = 0.01; high C: R^2 = 0.67, p= 0.09; Figure 2.6a). DO and TN were also related, but relationships were not as strong (low C: R^2 = 0.31, p = 0.33; med C: R^2 = 0.77, p = 0.05; high C: R^2 = 0.52, p = 0.17; Figure 2.6b).

TP concentrations within each reservoir ranged from 11 to 28 µg L^{-1}, 14 to 89 ug L^{-1}, 16 to 55 µg L^{-1} in the low, medium, and high carbon reservoirs, respectively. TP concentrations were lowest in the low carbon reservoir and highest in the medium carbon reservoir throughout all five flooding seasons (Figure 2.7). TP concentrations varied with depth in all three reservoirs after the first season of flooding. Bottom concentrations occasionally increased and were probably associated with temporary periods of stratification and low oxygen concentrations (Figure 2.5). Mean annual TN concentrations ranged from 341 to 429 μg L^{-1}, 399 to 702 μg L^{-1}, and 420 to 576 μg L^{-1} in the low, medium, and high carbon reservoirs, respectively. TN concentrations within reservoirs were highest during the first year of flooding and are equal to the maximum range values (Figure 2.8). TN concentrations were generally similar at all depths in all reservoirs with no depth having consistently higher TN concentrations.

N and P stored in periphyton at the end of each flooding season represents the net storage of N and P for the flooding season and ranged from 14 to 9519 g ha^{-1} for N and 1 to 681 g ha^{-1} for P, respectively (Figure 2.9). Periphyton was a substantial pool of N and P in all reservoirs in 2000, equaling about half of the annual N and P fluxes from reservoirs. N mass was greatest in the medium carbon reservoir (9519 g ha^{-1}), followed by the high carbon reservoir (5141 g ha^{-1}), and then the low carbon reservoir (3646 g ha^{-1}). Periphyton P mass followed the same pattern in 2000 with values of 681g ha^{-1}, 385 g ha^{-1}, and 295 g ha^{-1} in the medium, high, and low carbon reservoirs, respectively. In 2001, periphyton N and P masses decreased in all three reservoirs. Masses were largest in the high carbon reservoir (N: 2422 g ha^{-1}, P: 149 g ha^{-1}), smaller in the medium carbon reservoir (N: 899 g ha^{-1}, P: 76 g ha^{-1}), and smallest in the low carbon reservoir (N: 489 g ha^{-1}, P: 38 g ha^{-1}). In 2002, N masses decreased further and followed the same pattern, with the N mass increasing with site carbon content. In 2003, N increased in the low carbon reservoir (30 g ha^{-1}) and remained steady in the high (27 g ha^{-1}) and medium (22 g ha^{-1}) carbon reservoirs. P masses substantially decreased in 2002 and 2003 with a range of 1 to 3 g ha^{-1}. Periphyton N:P molar ratios were relatively consistent in all reservoirs from 2000-2002, with an average ratio of 31 ± 4 (Figure 2.10). In 2003, N:P increased to 53 in the high carbon reservoir and decreased to 24 in the low carbon reservoir, while N:P in the medium carbon reservoir slightly increased to 36.

Annual fluxes

Outputs of total nitrogen (TN) and total phosphorus (TP) exceeded inputs for all reservoirs in each year, generating positive net fluxes (Figure 2.11). TN fluxes were not significantly different in the three reservoirs (Friedman's test, $\chi^2 = 0.4$, df = 2, $P = 0.819$, $n = 5$) and neither were TP fluxes (Friedman's test, $\chi^2 = 1.2$, df = 2, $P = 0.549$, $n = 5$). TP fluxes were largest during the first year of flooding in 1999 with values of 2588 g ha^{-1}, 4157 g ha^{-1}, and 2003 g ha^{-1} in the low, medium, and high carbon reservoirs, respectively. TP fluxes tended to decrease after each subsequent flooding season, but reservoirs continued to be TP sources for five years. The lowest TP fluxes in the low (459 g ha^{-1}) and medium (675 g ha^{-1}) carbon reservoirs occurred during the last year of flooding in 2003. However, the lowest TP flux in the high carbon reservoir (603 g ha^{-1}) occurred in 2000. TN fluxes did not markedly decrease after the initial flooding season (Figure 2.11). There was no discernible pattern of TN flux in the low carbon reservoir (Figure 2.11). The largest TN fluxes in the medium carbon reservoir tended to occur in the first three years of flooding from 1999 to 2001 (27671 g ha^{-1}, 16571 g ha^{-1}, 17741 g ha^{-1}, respectively). In the high carbon reservoir, TN fluxes were largest in the last two years of flooding in 2002 to 2003 (14381 g ha^{-1}, 20099 g ha^{-1}, respectively). Neither TP nor TN fluxes were related to site carbon content (Figure 2.11), as we hypothesized. However, both TN and TP fluxes varied similarly in each reservoir (Figure 2.11). During five years of repeated flooding, the low, medium, and high carbon reservoirs released 73,303 g ha^{-1}, 81,293 g ha^{-1}, and 58,751 g ha-1 TN and 6,101 g ha^{-1}, 8,064 g ha^{-1}, and 5,961 g ha^{-1} TP, respectively.

Annual TDP and TDN fluxes were positive in all reservoirs throughout the experiment (Figure 2.11). TDN comprised the majority of TN in all reservoirs for all years. Therefore, patterns of TDN flux in each reservoir were similar to TN flux variation. PN fluxes were positive in all reservoirs during the first three years of flooding, but PN flux was negative in the medium carbon reservoir in 2002 (-459 g ha^{-1}, Figure 2.11). PN fluxes were largest in the medium carbon reservoir in 1999 (6527 g ha^{-1}) followed by the high carbon reservoir in 2002 (4584 g ha^{-1}) and 2003 (4241 g ha^{-1}). PP generally made up a larger proportion of TP than TDP. TDP fluxes were largest in the first year of flooding in 1999 in all reservoirs reaching 1483 g ha^{-1}, 2281 g ha^{-1}, and 843 g ha^{-1} in the low, medium, and high carbon reservoirs, respectively. PP fluxes were generally positive in all reservoirs for all years except for in the low carbon reservoir in 2003 (-23 g ha^{-1}, Figure 2.11). PP fluxes were largest during the first year of flooding in 1999. PP fluxes generally decreased after each flooding season in the low and medium carbon reservoirs but was more variable in the high carbon reservoir.

On average, molar TN:TP flux ratios were largest in the low carbon reservoir (34), followed by the medium carbon reservoir (27), and smallest in the high carbon reservoir (25, Figure 2.11). Paired t-tests showed no significant differences in TN:TP flux ratios among the three reservoirs (low carbon and medium carbon, $t = 0.95$, $df = 4$, $p = 0.40$; low and high carbon, $t = 1.93$, $df = 4$, $p = 0.12$; medium and high carbon, $t = 0.43$, $df = 4$, $p = 0.68$). TN:TP flux ratios were generally lowest in the first year of flooding in 1999 with values of 14, 15, and 12 in the low, medium, and high carbon reservoirs, respectively. TN:TP ratios generally increased with each flooding season and were mainly driven by decreases in TP fluxes since TN fluxes were relatively consistent among all years. In the low carbon reservoir, TN:TP increased in 2000 and 2001, but temporarily decreased again in 2002 (28). In the high carbon reservoir, TN:TP increased to 25 in 2000, but temporarily decreased again in 2001 to 16 (Figure 2.11). TN:TP in the medium carbon reservoir increased in 2000 and 2001, but decreased again in 2002 (26,

Figure 2.11). On average, TDN:TDP was smallest in the high carbon reservoir (42), followed closely by the medium carbon reservoir (43), and largest in the low carbon reservoir (50). TDN:TDP was smallest during the first year of flooding in all reservoirs (low: 20, med: 18, high:16). There was no clear pattern of TDN:TDP increase or decrease in any of the reservoirs. The largest TDN:TDP ratios occurred in 2001 in the low and medium carbon reservoirs (80 and 63, respectively) and in 2000 in the high carbon reservoir (60). Average PN:PP was largest in the high carbon reservoir (13), followed by the low carbon reservoir (9), and smallest in the medium carbon reservoir (6). PN:PP fluctuated from year to year in each of the reservoirs and there was generally no discernible pattern.

Discussion

Elevated N and P concentrations during flooding are often attributed to anthropogenic sources in runoff such as from agriculture fields of impervious areas. Here we find runoff from inundated natural areas is also a source of N and P. These naturally forested systems contributed N and P during flooding, especially when newly flooded. With repeated flooding of the FLUDEX reservoirs, the release of P declined, but the production of N continued thus dramatically altering N:P ratios. Elevated N and P were observed following flooding and the processes controlling these increases in nutrients has been attributed to organic matter decomposition or to the release from saturated soils (Kim et al. 2014). We took advantage of the controlled hydrology and frequent chemistry sampling in this experiment to quantify N and P release from organic matter under flooded conditions. We anticipated that flooding would increase TN and TP concentrations in floodwater. Indeed, TN and TP budget calculations show that the reservoirs were net sources of TN and TP for at least five years of repeated flooding. TN and TP flux magnitudes varied in each site in each year and were not directly related to site carbon content, therefore, leading us to explore the causes of these differences.

Increased TN and TP concentrations were observed in all reservoirs within one week of flooding. The role of increased moisture in stimulating microbial metabolism and organic matter decomposition as has been observed in flooded boreal forest soils and litters (Kim et al. 2014; Hall and St. Louis, 2004). Decomposition was likely the most important process driving N and P release since average TN and TP concentrations were generally related to average DO concentrations (Figure 2.6). Additionally, previous studies that quantified by-products of decomposition such as CO_2 and CH_4 showed that the three reservoirs were sources of these gases for all five years of flooding (Matthews et al. 2005, Venkiteswaran et al. 2013). Overall, internal TN and TP concentrations were not related to site carbon content indicating that the total amount of organic matter substrate was not the main factor controlling TN and TP release in the first 5 years of flooding. However, N and P release may persist longer in sites with more stored organic carbon. Generally, temperature and oxygen availability also regulate decomposition. However, temperature likely does not explain the variation in N and P production among sites since reservoir water temperatures were within 1.5ºC of each-other throughout the flooding experiment. Dissolved oxygen concentrations at the onset of flooding were high enough to support aerobic decomposition in all three reservoirs (Figure 2.5). Therefore, the difference in TN and TP production in the three reservoirs may have been dependent on dominant litter type in each reservoir instead of oxygen and temperature dynamics since they were similar in all reservoirs. A previous decomposition study noted that different litters present in these reservoirs decomposed at different rates even though internal conditions were similar (Hall and St. Louis, 2004). Therefore, organic matter quality and dissolved oxygen dynamics likely contributed to the variation in TN and TP production among reservoirs.

TN and TP fluxes were uncoupled over the five years of flooding, indicating that N and P release was not controlled solely by decomposition. TP fluxes substantially declined after the first flooding season, but TN fluxes remained elevated throughout all flooding seasons. DO, a measure of decomposition, was related to both N and P concentrations in all three reservoirs (Figure 2.6a-b). However, the relationship between DO and TP concentrations was stronger than between DO and TN concentrations. Differences in the amount of N and P release from flooded organic matter may also be related to uncoupled mineralization dynamics. There is some evidence from terrestrial N and P mineralization studies that P is preferentially mineralized over N (Marklein et al. 2015), which may have caused P to be depleted quickly and led to decreased P fluxes after the first year of flooding. Microbes mineralizing P can better detect changes in nutrient content and quickly deplete P stored in organic matter (i.e. within the first flooding season). N mineralizers, however, are more diverse and have different mechanisms for carrying out N mineralization causing N to be mineralized more slowly (Sinsabaugh et al. 2008). Additionally, N acquisition is related to organic matter quality further complicating N processing (Manzoni et al. 2008). Another explanation may be that there is simply more N stored in organic matter than P and the total amount of N and P released is dependent on the amount stored in organic matter. N release was sustained for at least 5 years, while P release decreased with each flooding season. Therefore, the relative amounts of N and P released from flooding may not be equal and changes in N:P ratios in reservoir outflows can be expected.

Flooding substantially altered N:P ratios in water leaving the reservoirs. TN:TP ratios were low in reservoir outflows in the first flooding season but tended to increase after each flooding season (Figure 2.3, 2.9). Outflow TN:TP was generally lower than inflow TN:TP indicating that these reservoirs were a greater relative source of P than N, but this is likely temporary and older reservoirs tend to act as P sinks (Maavara et al 2015). If N and P sequestered in periphyton were included in annual fluxes, then the initial decrease in N:P ratio at the onset of flooding in 1999 would have been even more pronounced. Previous studies have also shown that N:P ratios in reservoir outflows tend to be higher than inflows in older reservoirs (Cook et al 2010, Grantz et al 2014). In an ordinary reservoir system, outflow water would continue downstream where it could lead to changes in community composition and biogeochemical cycling (Cross et al 2007). Additionally, large volumes of water leaving reservoirs may be the primary inflow to downstream systems and can, therefore, drastically change downstream N:P ratios.

Flood-induced nutrient loading is generally attributed to anthropogenic nutrient sources such as agriculture and industry, but our results indicate that flooding natural areas can also contribute to nutrient loading. This nutrient source is especially important in areas where aquatic ecosystems tend to have low N and P concentrations. Nutrients mobilized during flooding can be an important biogeochemical component of aquatic ecosystems and support primary productivity, but in some areas, these large pulses of N and P may accumulate and contribute to algal blooms downstream of flooded sites (e.g. Lake Erie). In addition, nutrient enriched floodwaters, whether from natural or artificial flooding, can change N:P ratios in receiving ecosystems and cause changes in primary producer communities (Smith 1983).

Conclusion

Sustained nitrogen release after multiple floods suggests that organic matter can be a continuous source of nutrients in aquatic ecosystems for at least five years. Many other studies also report increases in N and P transport after flooding (e.g Carpenter et al. 2017; Hubbard et al. 2011). Increases in nutrients following flooding have been attributed mostly to anthropogenic sources in runoff, but we demonstrate that N and P release during flooding can occur in flooding natural areas. Therefore, headwaters can be a significant source of N and P during flooding and can potentially accumulate downstream further elevating nutrient concentrations within aquatic ecosystems. However, it is important to note that N and P release under flooding conditions was uncoupled to some degree. Decomposition was not the only process regulating N and P release. Differences in N and P mineralization processes may have caused P to be released faster than N, quickly depleting the P stores and allowing sustained N release for five years of repeated flooding. Preferential phosphorus mobilization suggests that P release during flooding can be immediate and of substantial concentration. Significant amounts of P released during the onset of flooding may be more problematic than N release, especially since systems are often P limited. Nutrient enrichment and its potential downstream accumulation resulting from flooding and high precipitation events can result in undesirable algal blooms, such as those occurring in Lake Winnipeg (McCullough et al. 2012) and Erie (Michalak et al. 2013). Future research should focus on the dynamics of N and P release in floodwaters to better predict large loads that may have detrimental impacts such as stimulating algal blooms or impairing water quality.

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Table 2.1. Annual water budgets for low, medium, and high carbon reservoirs from 1999-2003 in m^3.

	high carbon					medium carbon					low carbon				
Year	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003
inputs															
Inflow	78592	93260	84834	80532	88492	76749	69612	74671	51443	56903	96897	108579	106845	91011	115166
Precipitation	3070	4292	2996	2992	2583	2075	2899	2004	2022	1790	2614	3652	2525	2539	2255
Runoff	3893	10787	3688	5599	628	604	1654	565	858	111	76	209	70	101	14
Storage	69	69	69	69	69	0	0	0	0	0	142	142	142	142	142
Total	85556	108339	91518	89123	91702	79427	74164	77240	54323	58804	99587	112441	109440	93651	117435
Outputs															
Weir	60195	80214	63392	63193	60299	60756	57249	63037	44340	44479	33515	42517	50518	35243	43839
Drain	6801	6801	6801	6801	6801	4266	4266	4266	4266	4266	6975	6975	6975	6975	6975
Evaporation	2165	2331	2313	1980	2202	1475	1575	1550	1338	1500	1859	1985	1922	1622	1906
Seepage	16327	18924	18944	17080	22332	12929	11074	8386	4379	8559	57097	60822	49883	49669	64573
Total	85487	108270	91449	89054	91633	79427	74164	77240	54323	58804	99445	112299	109298	93509	117293

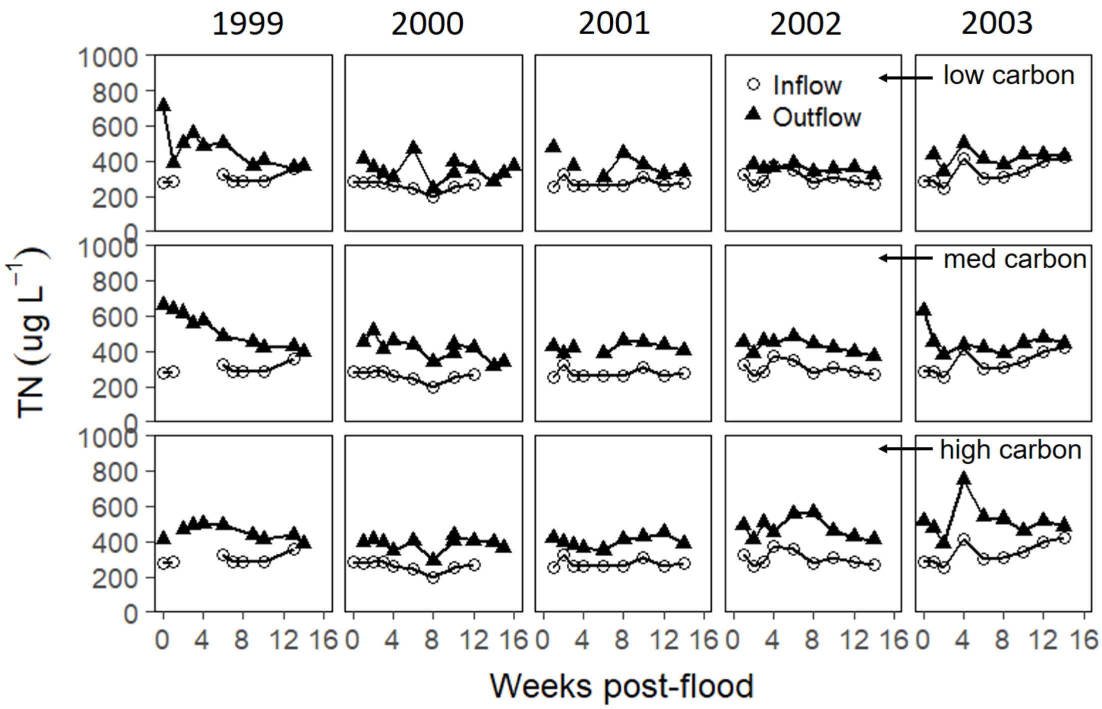

Figure 2.1. Total nitrogen (TN) concentrations in water pumped into reservoirs and in water leaving the low (top row), medium (middle row), and high carbon (bottom row) reservoirs over weirs during flooding from June to September in 1999 to 2003.

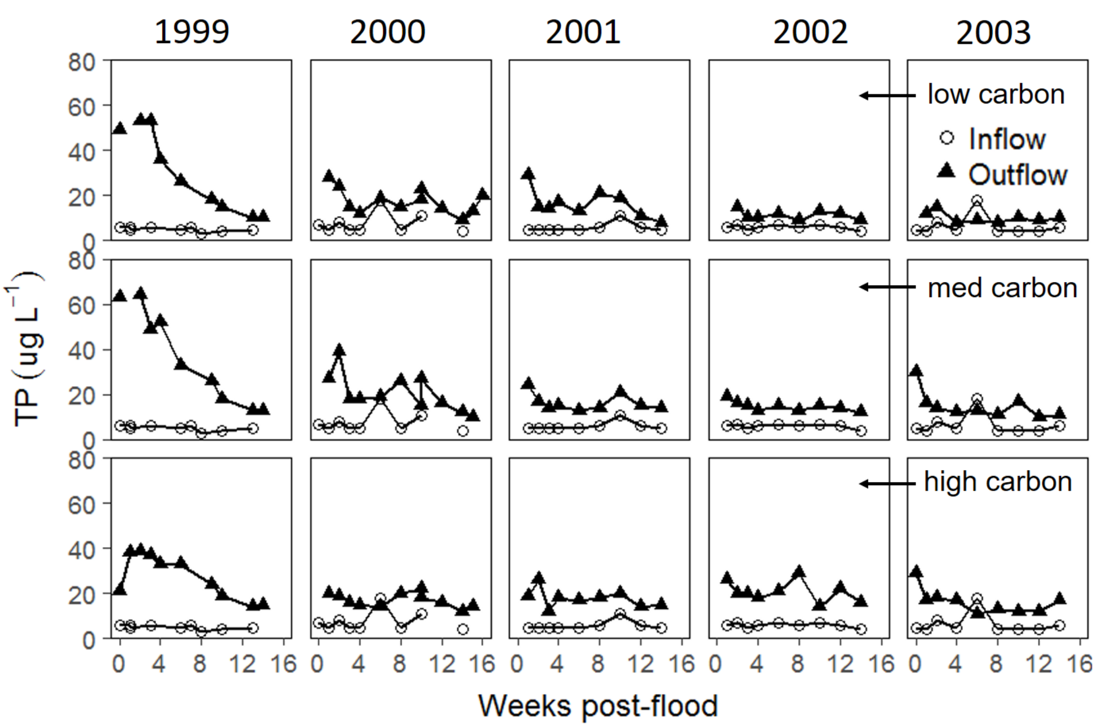

Figure 2.2. Total phosphorus (TP) concentrations in water pumped into reservoirs and in water leaving the low (top row), medium (middle row), and high carbon (bottom row) reservoirs over weirs from June to September in 1999 to 2003.

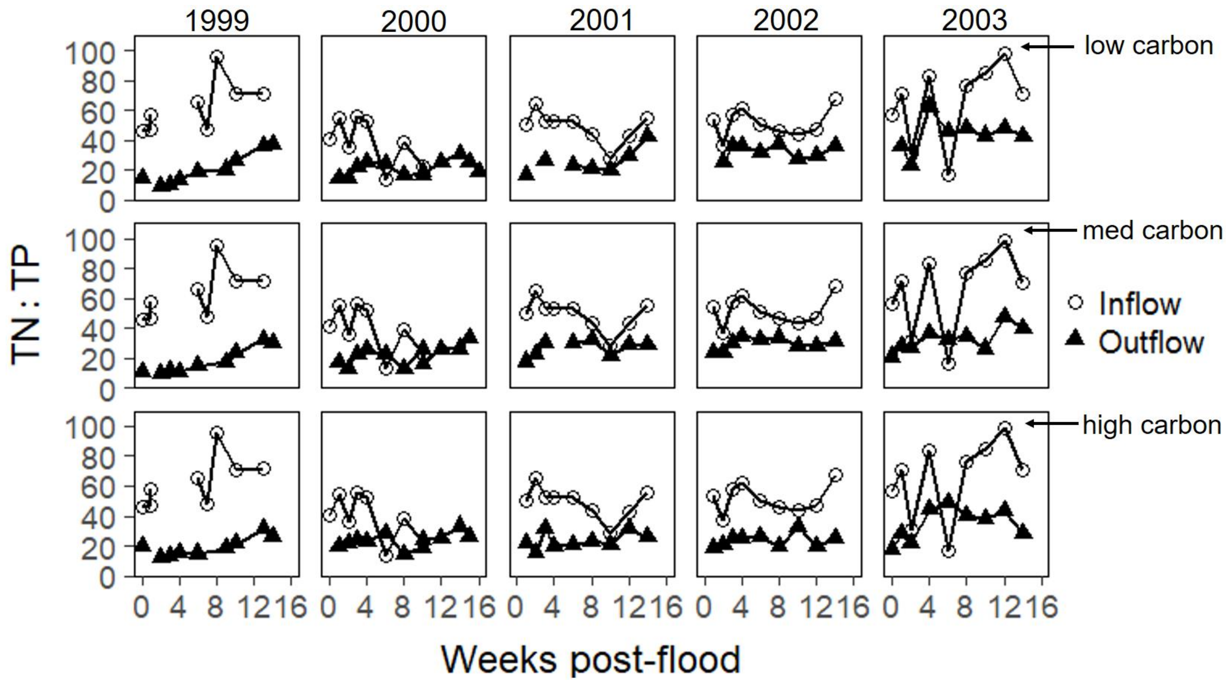

Figure 2.3. TN:TP ratios in reservoir inflow and outflow from June to September in 1999 to 2003 in the low (top row), medium (middle row), and high carbon (bottom row) reservoirs.

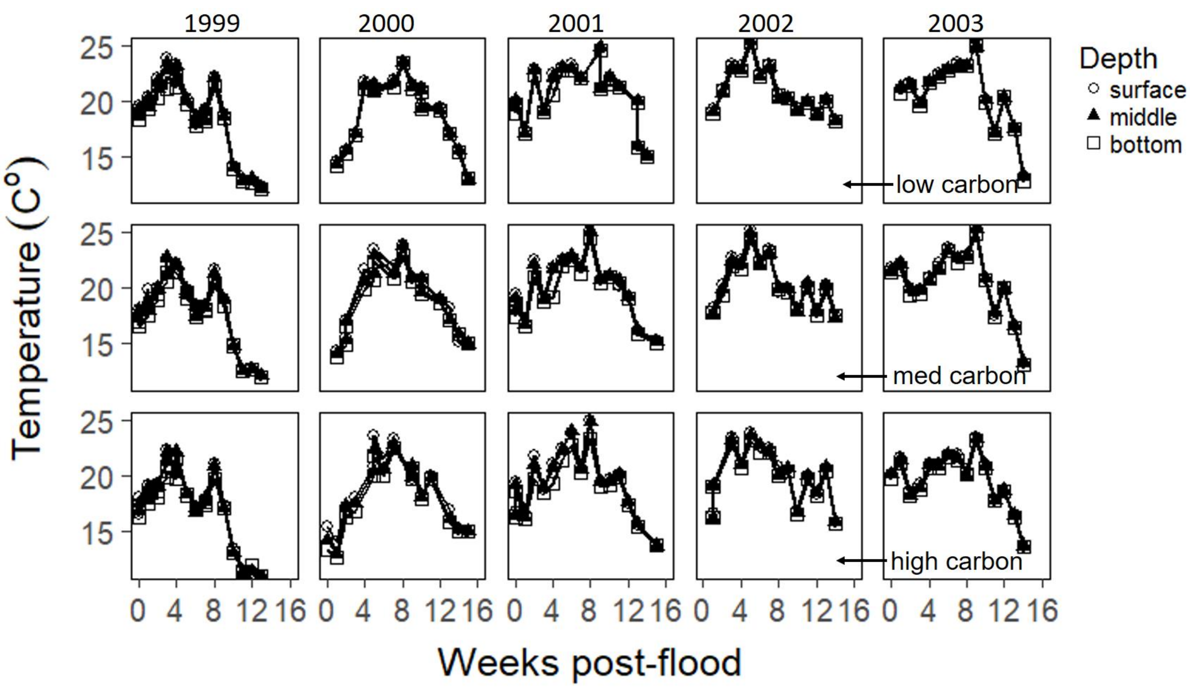

Figure 2.4. Temperature measured at the surface, middle, and bottom of the low, medium, and high carbon reservoirs from 1999 to 2003.

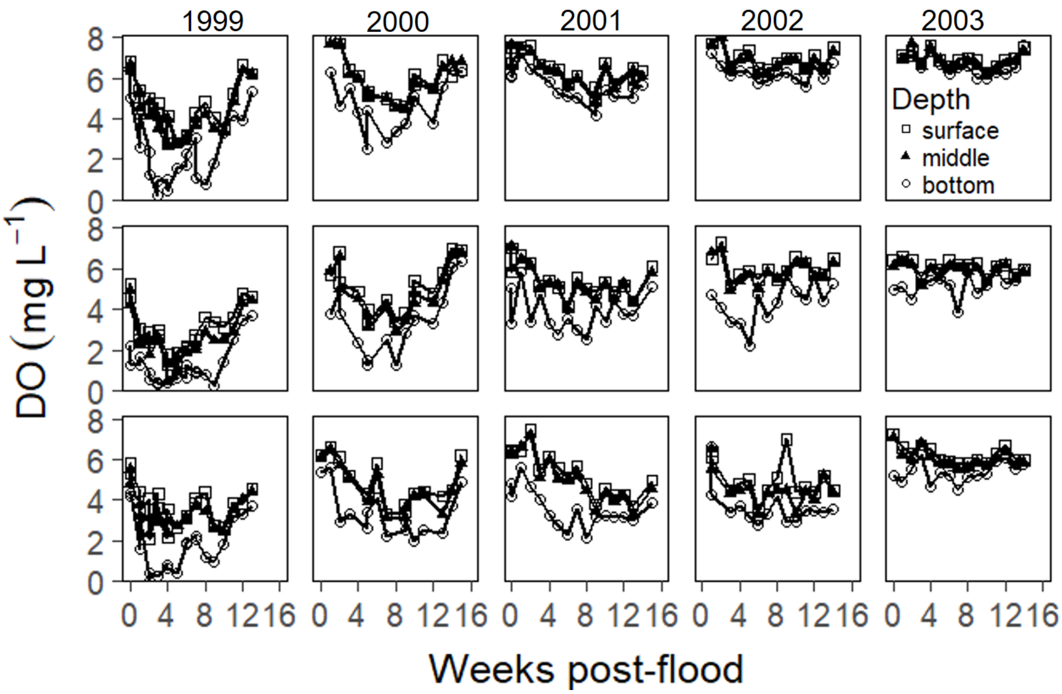

Figure 2.5. Dissolved oxygen (DO) concentrations measured at the surface, middle, and bottom of the low (top row), medium (middle row), and high carbon (bottom row) reservoirs during flooding from June through September 1999 to 2003.

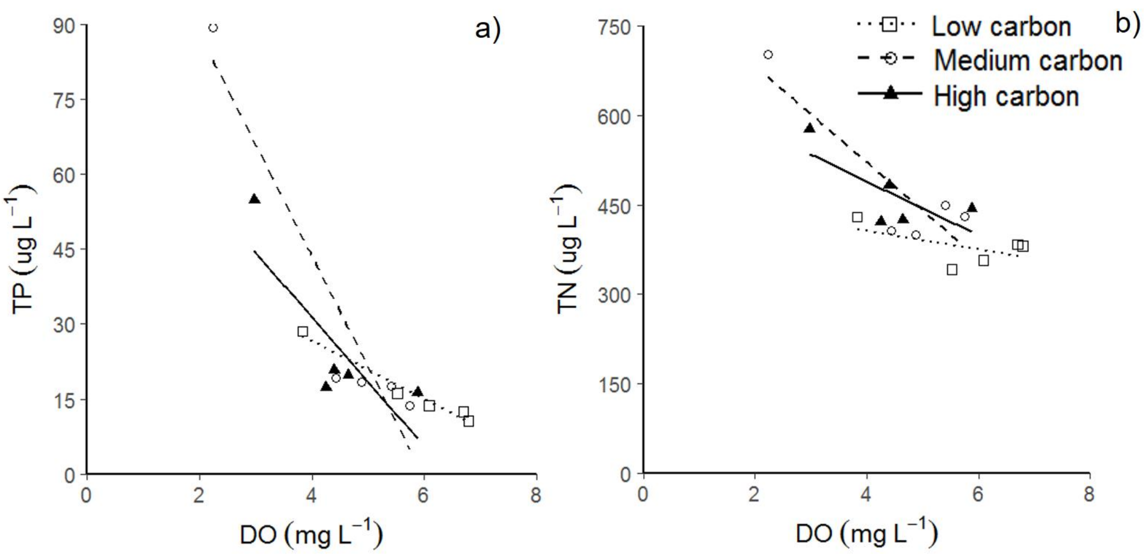

Figure 2.6. Linear regressions between TP and DO (a) and TN and DO (b) average annual internal reservoir concentrations in the low, medium, and high carbon reservoirs.

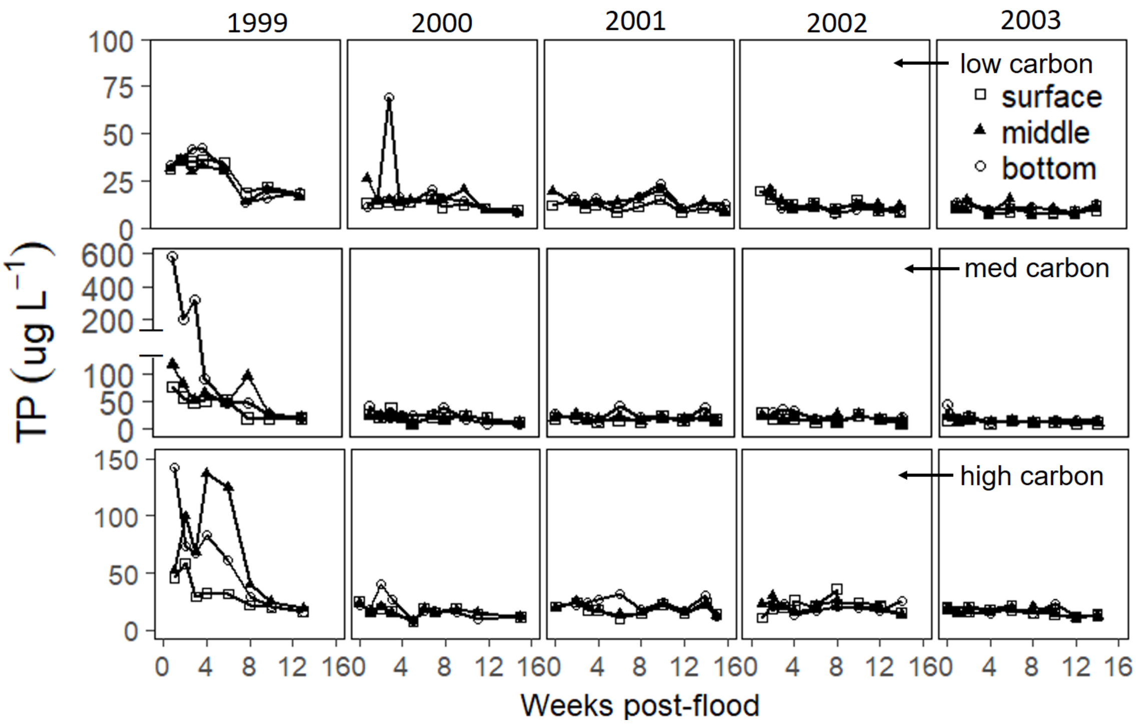

Figure 2.7. Total phosphorus (TP) concentrations measured at the surface, middle, and bottom of the low (top row), medium (middle row), and high carbon (bottom row) reservoirs during flooding from June through September 1999 to 2003.

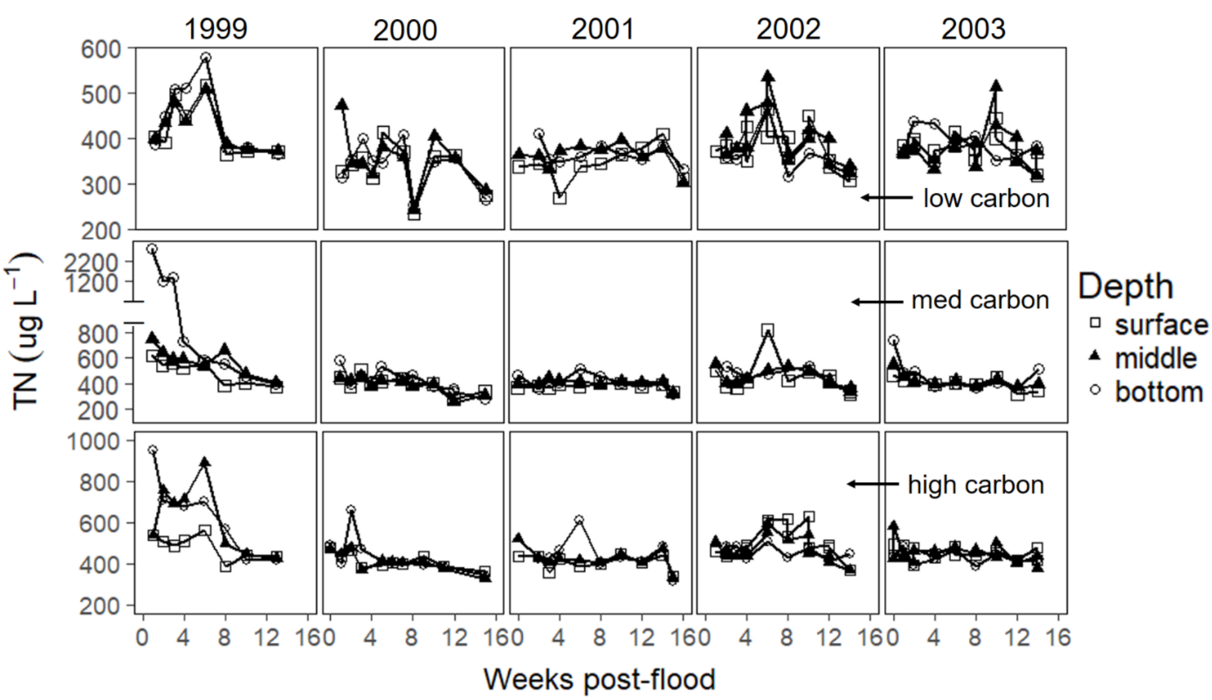

Figure 2.8. Total nitrogen (TN) concentrations measured at the surface, middle, and bottom of the low (top row), medium (middle row), and high carbon (bottom row) reservoirs during flooding in 1999 through 2003.

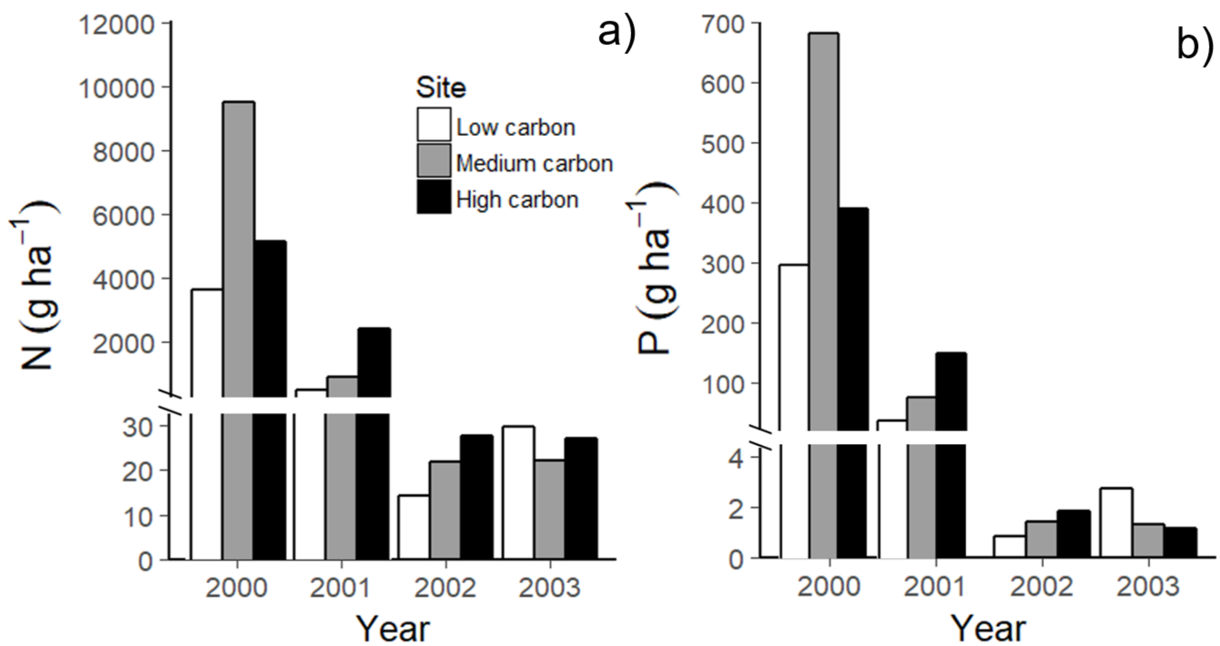

Figure 2.9. N (a) and P (b) stored in periphyton in the low, medium, and high carbon reservoirs at the end of each flooding season from 2000 to 2003.

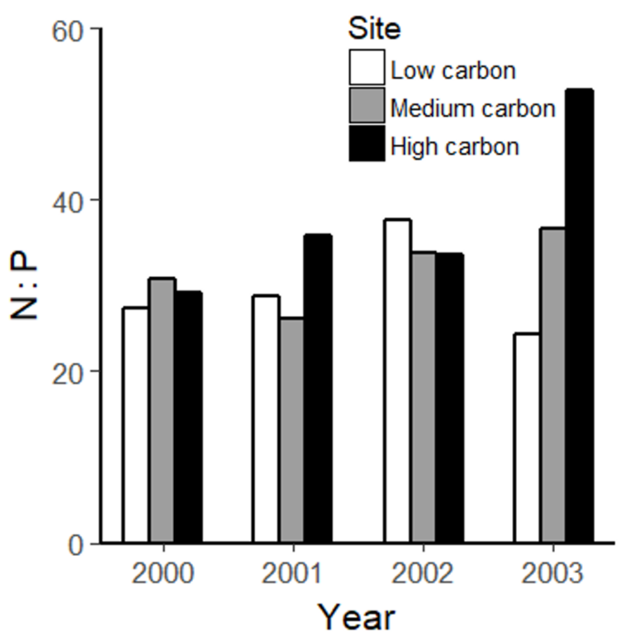

Figure 2.10. Molar N:P ratio in periphyton at the end of each flooding season from 2000 to 2003 in the low, medium, and high carbon reservoirs.

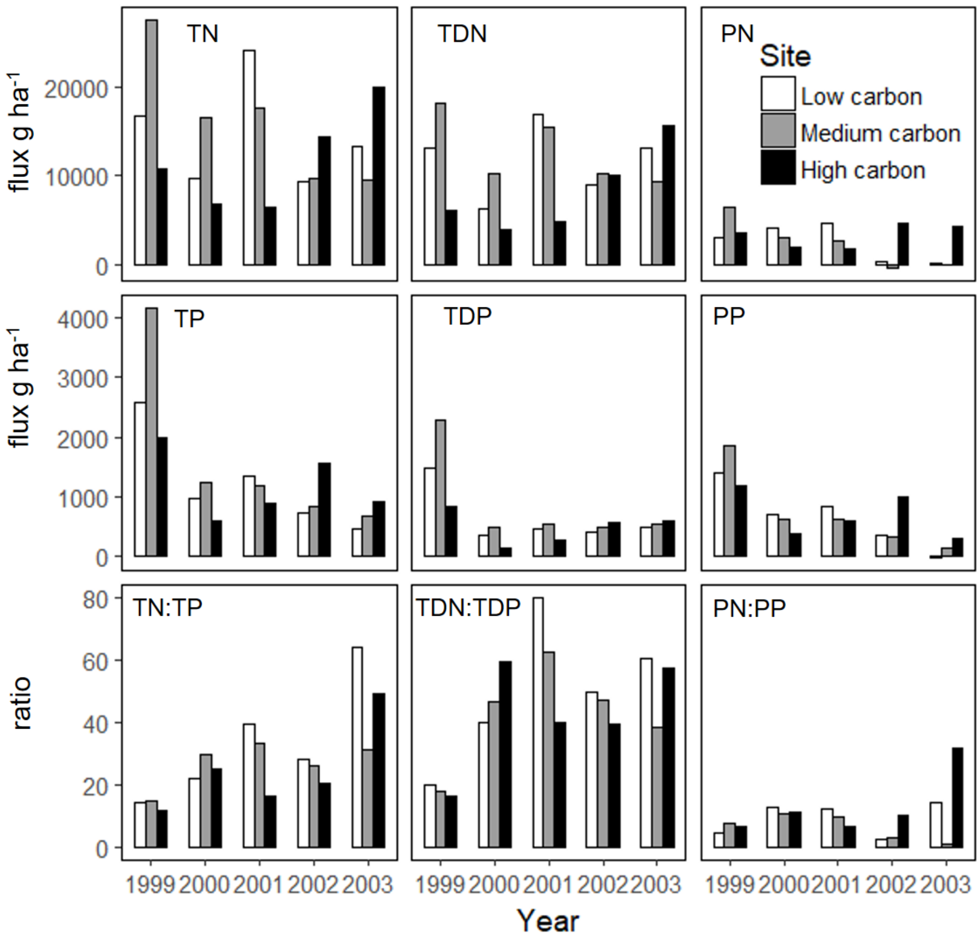

Figure 2.11. Calculated annual fluxes of total nitrogen (TN), total dissolved nitrogen (TDN), particulate nitrogen (PN) (top row), total phosphorus (TP), total dissolved phosphorus (TDP), and particulate phosphorus (PP) (middle row) in the low, medium, and high carbon reservoirs. Molar ratios of annual TN:TP, TDN :TDP, and PN:PP fluxes are in the bottom row.

3. The impacts of flooding on aquatic ecosystem services

Introduction

Flooding is usually considered a significant natural hazard causing disease, damage and loss to life, property, and infrastructure as well as disruption of public services. For example, floods can cause dangerous landslides (Hong et al. 2007), loss of crops and livestock (Atta-ur-Rahman and Khan 2011), disruption of normal drainage systems (Ogden et al. 2011), spillage of raw sewage and animal waste, and accelerated discharge of industrial and urban toxic materials (Euripidou and Murray 2004) and nutrients into waterways (Hubbard et al. 2011). Because of their dramatic effects on people and infrastructure, the effects of flooding on aquatic ecosystems are often viewed as negative; however, this is not always the case. Flooding can also provide many benefits, including recharging groundwater, increasing fish production, creating wildlife habitat, recharging wetlands, constructing floodplains, and rejuvenating soil fertility (Poff 2002). Since the effects of flooding on aquatic ecosystems can be both negative and positive, ecosystem services should also exhibit a mix of negative and positive outcomes resulting from flooding (Terrado et al. 2013). However, it is still unclear how floods of different magnitudes could affect gains or losses in ecosystem services ("the benefits people obtain from ecosystems" MA 2005) or how individual ecosystem services will be affected.

Floods occur when low-lying areas that are typically dry become temporarily inundated with water outside of their normal confines (Rojas et al. 2013). Flooding accounts for one third of natural disasters and affects more people than any other type of disaster (Sivakumar et al. 2011). Flood-related impacts are expected to worsen due to global environmental change with flood risk increasing by 187% from increasing temperature in the HadCM3 climate model (Arnell and Gosling 2016). Flood magnitude is also expected to increase due to intensified water cycling resulting from as little as a 1.5ºC global average temperature increase (Alfieri et al.

2017). However, all floods are not created equal and the causes and consequences of individual floods are often unique. Floods can be seasonal as in the case of spring snowmelt or monsoon rains or they can occur randomly via several other mechanisms such as ice jams, storm surges, and heavy precipitation (Figure 3.2a-c). Heavy precipitation accounts for about 65% of river floods (Douben 2006), but northern latitude areas with snow cover are also vulnerable to flooding caused by snowmelt and sometimes exacerbated by rain events (Kundzewicz et al. 2014). Flood events have been further characterized based on magnitude, frequency, duration, and volume (Burn and Whitfield 2016). These characteristics are important for determining the effects of floods on both aquatic ecosystems and the people who benefit from them. For example, flood magnitude can determine the amount of groundwater recharge or the extent of home and infrastructure damage during flooding. Flood magnitude is only one aspect of predicting flood impacts on aquatic ecosystems and ecosystem services. Ecosystem conditions prior to flooding are potentially equally as important as flood characteristics for determining ecosystem response to a flood event.

Rivers need floods to create unique habitat and support biological productivity and biodiversity. The Flood Pulse Concept states that predictable seasonal floods are beneficial for riverine systems and can influence biotic composition, nutrient transport, and sediment distribution but unpredictable floods may be disruptive for aquatic organisms (Junk et al. 1989). Additionally, many aquatic ecosystems have reduced resilience to future extreme events such as flooding due to human activities that include urban development and farming on floodplains, river flow disruptions, and pollution (Woodward et al. 2016). These activities increase the likelihood that floods become catastrophic events especially from the perspective of “benefits” obtained from ecosystems. The specific effects of flooding on aquatic ecosystems and their services are not well understood, but the importance of flooding for maintaining ecological functions in rivers has been recognized (Peters et al. 2016). Most of the research on flooding takes advantage of fortuitous events and thus often lacks pre-flood reference data (Poff and Zimmerman 2010). This relatively sparse evidence on how flooding and changes in hydrology impact aquatic ecosystems drives a large amount of environmental flow management (Acreman et al. 2014) and flood-related research.

Using an ecosystem service approach can help advance our understanding of the impacts of flooding on aquatic ecosystems and how future changes in flood magnitude will change the availability of aquatic ecosystem services. People have taken advantage of various ecosystem services for over 10,000 years (Fisher et al. 2008), making them integral to society. In fact, the estimated global value of all ecosystem services in 2011 was $125 trillion/year (Costanza et al. 2014). There are many studies that evaluate the effects of disturbances on ecosystem services, but most of these studies focus on terrestrial systems and there are few that look at aquatic ecosystem services (Grizzetti et al. 2016). Furthermore, there are even fewer studies that integrate the effects of hydrologic changes (Terredo et al. 2013). Aquatic ecosystems provide many services such as drinking water, soil formation, primary production, and areas for recreation or tourism, but flooding can impact the availability of these services. We expected to find that flood magnitude plays a role in determining whether aquatic ecosystem services are lost or gained following flood events. We expected that small floods would lead to gains in aquatic ecosystem services, while extreme floods would lead to losses. If ecosystem services respond to small and extreme magnitude floods differently, then current flood mitigation strategies may be detrimental to aquatic ecosystem services. Common flood mitigation activities such as damming and flood barrier implementation restrict the occurrence of small floods but are often unable to mitigate extreme floods (Alfieri et al. 2016).

In this study, we examined the societal pros and cons of various flooding events by evaluating their effects on aquatic ecosystem services. We used our current understanding of ecosystem services and flood impacts on aquatic ecosystems to identify gains and losses in ecosystem services resulting from flood events of different magnitudes. We completed a systematic literature review on a subset of 10 aquatic ecosystem services thought to be directly influenced by flooding to determine whether small versus extreme floods cause gains or losses in these services (Table 3.1). The ecosystem services included represent a variety of service types (i.e., provisioning, supporting, cultural, and regulating) from the Millennium Ecosystem Assessment framework (MA 2005) to create a holistic view of the ecosystem response to flooding. We also compared the influences of small versus extreme magnitude floods on each of the 10 ecosystem services to distinguish between normal (often seasonal) flooding and rare extreme events that may impact aquatic ecosystems differently. We hypothesized that small floods would enhance ecosystem service provisioning compared to large floods, which we expected would have more negative effects on ecosystem services. Ultimately, our study can be used to inform effective flood protection strategies that can mitigate the undesirable consequences of flooding while preserving aquatic ecosystem services. Decision makers may use the demonstrated importance of small versus extreme floods for ecosystem services to better manage for variable flows, including small and occasional extreme floods. Because ecosystem services are derived from well-functioning ecosystems, managing for ecosystem services may simultaneously benefit people and aquatic ecosystems.

Methodology

The Millennium Ecosystem Assessment (MA) aimed to address how ecosystem change can affect ecosystem services and their beneficiaries and to find a scientific way to ensure sustainable use and conservation of these services (MA 2005). Many ecosystem service frameworks have been developed since the MA such as Final Ecosystem Goods and Services Classification System (FEGS-CS; Landers and Nahlik, 2013), Stressor–Ecological Production function–final ecosystem Services (STEPS; Bell et al. 2017), and Ecosystem Service Profile (ESP; Paetzold et al. 2010). These frameworks and others typically focus on final services (services that people use directly) and emphasize economic valuation, which was not the goal of our analysis. Additionally, none of these frameworks are widely used (Nahlik et al. 2012). Therefore, we chose to use the MA framework to structure our analysis because it is commonly used to evaluate ecosystem services and is flexible enough to capture many types of services. We used a group of 10 ecosystem services identified by the MA framework spanning the following four MA categories; 1) regulating services (benefits resulting from the regulation of ecosystem processes), 2) provisioning services (services that provide a product), 3) supporting services (services that aid in the production of all other ecosystem services), and 4) cultural services (nonmaterial benefits) (MA 2005) (Table 3.1). Supporting services are ecosystem functions and processes, which aid in the production of other services (Brauman et al. 2007). For example, soil formation provides one of the materials necessary for agriculture, contributing to the provisioning service of food supply. Since the MA was completed, the ecosystem services concept has evolved and supporting services are now typically considered ecosystem functions rather than benefits or ecosystem services (Haines-Young and Potschin 2010). However, we included supporting services in our analysis in order to capture a larger range of possible aquatic ecosystem responses to flooding. In contrast, provisioning services provide a material product that can be harvested or collected and then traded in markets (Brauman et al. 2007). Regulating services regulate ecosystem processes, providing a suitable environment for people to live in (Braat and de Groot 2012). Cultural services are also non-material goods. They provide sensory experiences that enhance quality of life such as areas for recreation and tourism and aesthetic value. Ecosystem services can be assessed either by quantifying biophysical changes or by assigning a dollar value to those changes (Braat and de Groot 2012). We used indicators of ecosystem service changes derived from variables measured in studies collected during our literature review to determine gains and losses in ecosystem services after flooding. We found that a variety of indicators or variables were used to report changes in the same ecosystem service; therefore, we included as many commonly reported indicators as possible. Because each flooding event is context dependent (e.g., antecedent conditions, soil conditions, ambient water conditions, etc) and pre-flood data was often lacking from studies we could not quantify a general response to floods. Instead, we provide a *general pattern* (rather than a quantitative change) of ecosystem service changes in response to flooding.

We performed a systematic literature review to locate existing research on the effects of flooding on ecosystem services. We obtained published articles from Web of Science from 1980 to 2017 and summarized them. We focused upon the impacts of river basin flooding rather than flooding involving seawater intrusion or saltwater flooding, but studies included contained a variety of flood-generating mechanisms such as monsoons, cyclones, snowmelt, storm surges, and heavy precipitation. We chose to use flood return interval to characterize floods as either small or extreme because it is commonly present in the published literature. Other flood characteristics such as duration and frequency are also important for determining the effects of flooding but were rarely reported in published literature and therefore not explicitly considered in this study. We aimed to include both small floods (defined as < 10-year recurrence interval) and extreme floods (> 100-year return interval). This was a challenge because the impacts of small and seasonal floods are often not reported (Douben, 2006). Therefore, the analyses of extreme flood impacts on ecosystem services are more complete. We searched for each ecosystem service individually. Each search began with the terms "flood" OR "flooding" OR "floods". Then, specific terms related to each indicator were added. For example, the terms "("flood" OR "flooding" OR "floods") AND river AND ("outbreak risk" OR disease)" were used to search for literature relevant to human disease regulation. We followed-up the initial literature search with searches aimed at finding additional studies on small floods. We used the same ecosystem service-specific terms but replaced "flood" with "high discharge" and "storm". This increased the number of results returned during searches, but many studies were excluded because they did not report overbank flow or inundation, thus not allowing us to accurately characterize the flood. All studies with abstracts containing information about a specific flood or storm event and a variable representing an ecosystem service were downloaded. We screened each of these studies one additional time to identify studies, which included a quantitative measure of the flood impact such as before and after measures of the same variable (e.g. Table 3.2). These initial literature results were augmented by further targeted searches on specific services and other work cited in the initially identified papers.

This resulted in 117 studies after the literature search given described constraints. Each ecosystem service was represented by an average of 12 ± 4 studies. In general, the literature reported negative effects associated with flooding. Flooding is commonly perceived as detrimental and most studies tend to focus on the negative impacts of floods rather than the positive impacts. This bias may have skewed our results toward greater ecosystem service losses, but we were still able to identify ecosystem services which benefit from flooding. Ecosystem service availability varied with flood magnitude (Figure 3.1; Table 3.3). Both small and extreme floods generally decreased the availability of most ecosystem services. However, extreme floods caused a greater number of ecosystem service losses than small floods (Table 3.3). Extreme floods were beneficial for groundwater and aquifer recharge and therefore were positive for these services. Small floods were important for improving access to food and recreation as well as beneficial for water regulation and primary production. The impacts of floods on ecosystem services were also related to initial physical, chemical, and biological conditions within the ecosystem and its location. These complex interactions made it difficult to attribute changes in ecosystem services to specific flood events. For example, post-flood changes in primary production varied because of temperature, light, and nutrient conditions. Additionally, there was some variation within individual ecosystem services which made assigning a negative, neutral, or positive outcome difficult. However, we were able to identify many of the possible underlying mechanisms that were responsible for ecosystem service outcomes post-flood from reviewed literature (Figure 3.3). Below we describe each ecosystem service and its connection to flooding in more detail.

Supporting services

Primary production

Hydrology is known to influence primary production by affecting water clarity, oxygen, pH, and nutrient concentrations (Lindholm et al. 2007). Floods may initially inhibit primary production while water is high but nutrients mobilized during storms may be held and processed in ecosystems later, when water levels return to normal (Paerl et al. 2011). Small seasonal floods contribute nutrients to aquatic ecosystems and can stimulate primary production (Junk et al. 1989), a process that is especially important in nutrient-poor oligotrophic systems. Increased primary production can then support aquatic food webs, providing a food source for consumers (Alford and Walker, 2013). However, larger floods can transport excessive nutrients and potentially stimulate excessive primary production (i.e., eutrophication) or alter primary producer community composition, causing unfavorable species to dominate. Recently, increases in primary production have been attributed to increased phosphorus (P) and nitrogen (N) loading associated with flood events (Paerl et al. 2016). For example, flooding in the Lake Winnipeg catchment increased phytoplankton biomass and the phytoplankton community shifted to include more cyanobacteria (McCullough et al. 2012). Heavy rainfalls in the Lake Erie basin caused significant P loading and resulted in the largest algal bloom in the lake's history (King et al. 2017). Harmful algal blooms (HABs) such as those which occurred in Lakes Winnipeg and Erie cause several problems for people who rely on these water bodies for drinking water and recreation. HABs include cyanobacteria which produce toxins that must be removed from drinking water supplies (Hitzfeld et al. 2000). HABs also lead to poor aesthetics, which adversely affect tourism and recreation activities, with detrimental impacts on local economies such as those around Lake Erie (Watson et al. 2016). Primary production benefits aquatic ecosystems up to a certain tipping point, when HABs can dominate and negate these benefits (Paerl et al. 2016). Therefore, increased primary production post-flood is considered an ecosystem service net gain but if primary production is excessive then flooding results in a net loss. Additionally, if a flood event decreases primary production, then it is considered a net loss.

Our literature review uncovered no consistent patterns of post-flood primary production responses. Both increases and decreases in primary production after flooding were reported. One study reported higher gross primary productivity (GPP) after a small flood (e.g. Lindholm et al. 2007), but other studies reported lower GPP post-flood (e.g. Uehlinger 2000; Uehlinger et al.

2003). Chlorophyll *a* (used as a surrogate for primary production) concentrations were also observed as decreasing after small floods (e.g. Rodrigues et al. 2002; Weilhofer et al. 2008). Differential responses in primary production are likely the result of differences in nutrient supply, light penetration, and flushing rates of impacted ecosystems (Paerl et al. 2014a; 2014b; 2016). Additionally, post-flood increases in nutrient supply must occur simultaneously with sufficient light penetration to cause increases in primary production. Minor et al (2014) found that increases in post-flood P did not increase primary production because light was limited by increases in total suspended solids (TSS) and chromophoric dissolved organic matter (CDOM). The two studies reporting on the effects of extreme flooding on primary production also contained mixed results. Silva et al. (2013) reported that extreme flooding increased net primary productivity (NPP). The second study reported that chlorophyll *a* did not change after a "high magnitude" flood (Weilhofer et al. 2008). In addition to providing nutrients, freshwater discharge resulting from flood events modulates the rate of flushing (or water residence time) of receiving waters. If flushing rates exceed algal growth rates, large flood events could reduce algal biomass, regardless of nutrient enrichment (Peierls et al. 2012; Paerl et al. 2014b). We therefore cannot consistently conclude whether flooding increases or decreases primary production and algal biomass since these indicators are highly dependent on other, interacting variables such as nutrient enrichment, water clarity, flushing rates, and grazing. However, the potential for large algal blooms occurs after flooding when nutrients are high and water residence time is long enough to allow blooms to form and accumulate (Paerl et al. 2016).

Soil formation

Soil formation provides an essential service by regenerating river banks, wetlands, and flood-plain farmland. Flooding causes over bank flow and changes the rate of sediment deposition and erosional processes occurring between the river and floodplain (Junk et al. 1989).

Flooding can cause river bank erosion and collapse, as well as upland erosion and incision, leading to landslides in areas with hillslopes and mountainous terrain (Larsen and Montgomery 2012) which pose threats to people (e.g. Kala 2014). Alternatively, flooding can improve soil formation by depositing sediment on floodplains, which recharges farmland soils and increases suitability for farming (Ogbodo 2011). Therefore, the net positive or negative impacts of flooding on soil formation depend on where erosion and deposition occur and the volume of sediment transported.

The influence of a flood event on erosion and accumulation is related to the flow peak magnitude (Julian and Torres 2006). Extreme floods increase erosion, but up to 70% of eroded sediment can be re-deposited within the catchment (Morche et al. 2007). Such re-deposition events are important in maintaining coastal forests and wetlands (e.g. Nyman et al. 1995; Bryant and Chabreck 1998; Shaffer et al. 2016) that act as key buffers against storm surges, biogeochemical filters for water entering coastal oceans and large lake systems, and critical nursery sites for important fisheries (e.g. Barbier et al. 2011). Therefore, soil erosion processes are spatially dynamic and the negative effects of erosion in certain locations, such as river banks or hill slopes, may enhance soil formation in other areas of a catchment, such as floodplains (Pearson et al. 2016). Such effects can be strongly exacerbated by land use practices, and over time, can lead to both improved farming locations and detrimental, even catastrophic flooding within the same river basin, as illustrated by the Yellow River catchment in China over the past 7000 years (Rosen et al. 2015). We found that extreme flooding caused substantial amounts of soil to be eroded in all studies. In one study, the volume of soil eroded during an extreme flood was 87% of the total eroded volume during a period of six years (Carroll et al. 2004). Another study reported over 1.4 million m^3 of soil was eroded from a catchment in New Zealand (Fuller 2008).

Small floods also influence soil formation, although their effects are less dramatic than extreme events. Some studies, such as one by Dewan et al (2017), have shown that discharge and erosion are correlated so small floods likely cause a small amount of erosion. In addition to less erosion, small floods lead to less sediment accretion on river banks. Stromberg et al (1993) compared sediment accretion on banks following flood events with 2-year, 5-year, and 10-year flood recurrence intervals in Arizona, USA. They found that soil accretion generally increased with flood magnitude, but sediment accretion was similar in the 2-year and 5-year floods compared to the 10-year flood (Stromberg et al. 1993). Studies reporting the effects of multiple small events were more common than those reporting on single flood events. An example of a multiple-event study is by Leyland et al (2017), where they found that the mean rate of soil erosion was 4 times larger than the mean rate of soil accretion during the 2014 monsoon season in the Mekong River catchment. Multiple-event studies are difficult to compare because some include an entire flooding season, while others include a few flood events. Therefore, more studies on small individual flood events would be beneficial for assessing the impacts of small floods on soil formation.

Regulating services

Water regulation

Flooding is important for recharging underground water sources and recharge that results from flooding is especially beneficial during dry seasons when groundwater is the main source of freshwater in areas that experience pronounced wet and dry seasons (Kazama et al. 2007). In most cases, floodwaters are beneficial to recharge groundwater but this equation is changing with population growth. Demand for drinking water and water for irrigation will increase with population growth (Singh et al. 2014) and put further stress on surface water supplies that are already extensively exploited, causing people to rely more on groundwater (Wada at al. 2014; FAO 2016). As a result, human populations deplete underground water stores through extraction for irrigation and, to a lesser extent, drinking water. The need for irrigation to supply water to crops will also likely increase in areas where global environmental change is expected to increase temperatures and change precipitation patterns and where people are converting natural land covers to agricultural land (Taylor et al. 2013).

The effects of flooding on water regulation vary depending on floodplain conditions and natural hydrologic variability. For example, there is evidence that groundwater recharge is dependent on flood duration (Benito et al. 2010; Dahan et al. 2008) and floodplain land use (Keilholz et al. 2015). Additionally, inundation area determines how much floodwater infiltrates groundwater stores and larger inundation areas lead to more groundwater recharge. Therefore, flood mitigation strategies that reduce inundation area are detrimental to groundwater recharge processes (Kazama et al. 2007). However, groundwater levels that increase during flooding and extend above riverbeds or the soil surface can also contribute to more extreme flooding (e.g. Gotkowitz et al. 2014). Groundwater flooding can last longer than riverine overbank flooding and possibly inundate basements, agricultural land, and roads (Hughes et al. 2011). Therefore, it is optimal when groundwater is recharged but not to the point of overfilling during floods.

In our review of past flooding events, groundwater recharge increased with flooding in all 13 studies. Most studies reported that extreme floods contributed more water to underground stores than small floods, but one study showed that smaller floods contributed a disproportionately large amount of water to groundwater stores (Aksoy and Wittenberg 2015). Extreme floods contributed high volumes of water to groundwater stores. For example, an extreme flood increased the groundwater level by 0.8m, causing additional above ground flooding (Gotkowitz et al. 2014). Additionally, Wang et al. (2015) reported that an extreme flood event increased groundwater depth by 3.24 m. Small floods occurring seasonally were also capable of supplying substantial amounts of water. For example, one seasonal flood increased groundwater level by more than 0.5 m (Amiaz et al. 2011). In another study, spring flooding contributed 40% of water to the annual groundwater recharge (Ray et al. 2002). Therefore, both extreme, rare floods, and small floods occurring seasonally lead to increased water volume in underground water stores and improved water regulation.

Water quality

Flood events have contrasting effects on water quality. Increased terrestrial runoff from both surface and subsurface flow paths mobilize more dissolved nutrients on the landscape and reduce residence time in potential terrestrial sinks compared to water entering during base flow (Buda and Dewalle 2009; Bende-Michl et al. 2013). As a result, more nutrients are loaded into surface waters. However, while fluxes of dissolved constituents always increase during storms, concentrations show varied responses and may actually decline due in part to dilution during high flow events (Goodridge and Melack 2012; Carey et al. 2014; Wollheim et al. 2017). In contrast, sediment concentrations and dissolved organic matter concentrations generally increase during storms, so that fluxes will increase at greater rates than discharge (Raymond and Saiers 2010; Williams 1989). Total suspended solids (TSS) increases are further exacerbated in urban and agricultural catchments (Pizzaro et al. 2014), while dissolved organic carbon (DOC) tends to increase more in forests and wetland systems (Huntington and Aiken 2013). TSS and DOC have direct drinking water quality implications, while the impact of nutrients is often more indirect through ecosystem function such as stimulating primary production and creating suitable habitat and resources for aquatic organisms. Thus, extreme flood events are likely to exacerbate water quality issues, particularly in watersheds dominated by anthropogenic land uses.

Water quality is further influenced by transport, mixing, and dilution within the river network (Hale et al. 2014). As a result, the spatial pattern of water quality degradation depends on the extent of the extreme event relative to pollution sources, the amount of runoff from clean water generating regions, and their spatial connectivity, which is also a question of scale. For example, a pollution source located downstream may be considerably diluted during extreme events due to massive upstream water inputs, as is evident in the Merrimack R. watershed, New Hampshire, USA (Samal et al. 2017). Total flux still increases, but concentrations can decrease due to dilution, so water quality impacts will depend on whether total flux or concentrations are more important for determining effects of pollutant changes.

Finally, aquatic transformations within the river network may affect water quality. Transformations include retention (e.g., settling of sediments, assimilation of nutrients) or permanent removal (e.g., denitrification). This regulating ecosystem service is strongly affected by flow (Doyle 2005; Hale et al. 2014; Wollheim et al. 2008; Wollheim et al. This Issue). Generally, as flow increases, the ability to regulate downstream dissolved fluxes declines. However, this decline is a function of watershed size (length of flowpaths within a river network), the distribution of sources within the watershed, the abundance of lakes, reservoirs and wetlands, as well as connectivity with floodplains (Mineau et al. 2015; Wollheim et al. This Issue). Extreme floods are likely to connect flowing waters with floodplains where soils high in organic matter may remove nutrients (Ensign et al. 2008). Models suggest that there is an optimal level of inundation for nutrient removal at network scales, most likely when flood waters are shallow and widely dispersed, and before waters become deeper (with less contact with sediments) (Noe and Hupp 2009). However, this has not been empirically demonstrated. Nevertheless, floodplains are likely to regulate downstream fluxes where they occur. Anthropogenically-driven modifications such as levee building disconnect channels from floodplains, and thereby remove this function. As a result, storms transport more material downstream, potentially degrading water quality.

Regulation of human disease

Extreme flooding is a leading cause of weather related infectious disease outbreaks (Cann et al. 2013) and can overwhelm or damage sanitation systems, lowering the quality of water treatment, and in more extreme cases allowing sewage, industrial waste, and agricultural waste to mix with drinking water (Figure 3.2d). Increases in disease after floods range from waterborne infections such as cholera and hepatitis A, to pathogens with more complex life cycles and transmission pathways like schistosomiasis and malaria. Flooding can disproportionately affect populations that are already at increased risk of disease due to poverty, poor sanitation and housing, and limited access to healthcare systems. Quantifying disease occurrence attributable to floods is complicated by the long lag periods between the flood and disease presentation, as well as differences by location and population. Despite these difficulties, multiple studies have revealed associations between flooding and increases in disease.

Pathogen transmission can occur through ingestion of contaminated drinking water or direct contact with flood waters. Due to these mechanisms, diarrheal and gastrointestinal (GI) illnesses are among the more common diseases noted after floods. The relatively short lag period between flooding and increases in GI illness noted in multiple studies indicated a viral infection due to direct contact with contaminated flood water (Ding et al. 2013; Wade et al. 2004; Wade et al. 2014). Other viral GI pathogens such as norovirus have been linked to outbreaks due to direct contact with sewage contaminated flood waters (Schmid et al. 2005). Illnesses such as hepatitis

A, bacillary dysentery, and diarrhea were also hypothesized to be due to direct exposure to floodwaters or contaminated drinking water (Gao et al. 2016). A study of typhoid in Dhaka, Bangladesh showed that cases increase geographically around rivers and temporally after heightened rainfall and river levels (Dewan et al. 2013). Disease risk can also be modified by water source and possible disruption and changes in water source as a result of flooding. Kazama et al. (2012) showed risk of GI illness was inversely related to flood size in residential areas with smaller floods conferring greater risk than larger floods. The risk of infection was also mediated by water source, with greater risk from groundwater sources than surface water sources in sparsely populated regions (Kazama et al. 2012).

The effect of flooding on diarrheal illness is subject not only to the severity of the flood but the weather status prior to the flood. Heavy rainfall following dry periods could pose greater risk of diarrheal illness than continuous periods of wet weather (Carlton et al. 2014). A study of recurrent floods in India showed that long-term impacts of seasonal flooding are not as significant as that of sporadic flooding on childhood diarrheal illnesses (Joshi et al. 2011). It is possible that in contrast to sporadic flooding, seasonal floods are predictable dangers in some regions and preparations can be made to avoid related illnesses. Extreme flooding has been reported as a risk factor for cholera outbreaks in many regions as well (Griffith et al. 2006). Dual peaks in cholera occurrence in the Bengal delta were explained by both droughts and floods in the region (Akanda et al. 2009). Two studies following illness after consecutive major floods in Bangladesh showed variation in the causative pathogens of diarrhea by flood with the most common pathogen being *Vibrio cholerae* followed by rotavirus. Differences among the floods could be due to the natural seasonality of the diseases and other secular trends in healthcare occurring at the time of flood (Harris et al. 1998; Schwartz et al. 2006).

Incidences of disease which occur after flooding may be contracted through routes of exposure besides drinking water such as direct contact with floodwaters, where pathogens can enter the body through exposed or broken skin. A study of the health effects associated with the 2013 Alberta (Canada) floods revealed increases in tetanus shots and injuries associated with flooding (Sahni et al. 2016). Depending on the setting and the ability of the population to avoid the inundated area during the flood, it is possible that the majority of this direct contact risk comes from the clean-up process and not the initial inundation phase of the flood (Fewtrell et al. 2011). Direct exposure to flood waters can also lead to outbreaks in certain zoonotic disease such as leptospirosis in endemic Southeast Asian and south/central American countries, with municipalities lying in floodplains often correlated with higher rates of disease (Barcellos and Sabroza 2001; de Resende Londe et al. 2016).

Floods can also indirectly impact human health by supporting or spreading breeding grounds and dispersal of pathogen vectors. Flooding along the Yangtze River, China corresponded with the spread of schistosomiasis carrying snails to previously disease-free areas. Cases of schistosomiasis among humans and animals rose after a large flood in the area and the highest rates were localized to lakeside provinces along the Yangtze (Wu et al. 2008). Malaria was found to increase after extreme flooding in multiple studies due to the creation of stagnant pools of water that are necessary breeding grounds for the mosquitoes that carry and spread the pathogen. Boyce et al. (2016) showed malaria rates increased by 30% in areas bordering a recently flooded river. This spike in morbidity occurred at a time that was uncharacteristic for malaria season and was attributed to the flood waters creating stagnant waters for breeding that otherwise would not be present (Boyce et al. 2016). A temporal analysis of malaria after extreme flooding showed peak malaria rates at 25 days post-flood, consistent with the delay expected for mosquito growth, disease transmission and presentation (Ding et al. 2014). This lag period is much longer than that associated with viral GI illness and raises the issue of identifying an appropriate surveillance period when monitoring flood-related disease outbreaks. For certain diseases, a flood-related event might not show increases in cases until weeks after the flood has receded, especially if the organisms are able to remain in the soil. An outbreak of cryptosporidium among children in Halle, Germany was linked to their participation in activities on a floodplain two weeks after flood waters had receded and the floodplain had been reopened to the public (Gertler et al. 2015).

It is clear that flooding has important impacts on infectious disease but future research is needed on the relationship between flood size, flood occurrence, environmental conditions, and risk of health impacts. Unfortunately, many other methodological issues continue to complicate our understanding of the links between flood events and disease. Improved disease surveillance and flooding impact assessments need to be made, with better record keeping and sharing between government, relief, and other agencies involved in flood response. The disruptive nature of flood events can limit access to hospitals, possibly resulting in underestimates of disease rates if using hospital admission data or other forms of passive surveillance. Certain disease outcomes such as GI illness often may not require an ER visit or hospitalization which could also lead to underestimates of disease rates after flooding. Studies are also often correlative. Correlation analyses could be exposing direct relationships between flooding and disease or possible indirect relationships due to associations between flood risk areas and susceptible or high-risk populations. Extreme weather events convey a risk with respect to waterborne diseases and will disproportionately impact sectors of populations with preexisting health problems (Cann et al. 2013) and which lack preparedness (Sahni et al. 2016). Very large floods can also act to concentrate the population in areas with polluted water and poor hygiene services (Griffith et al. 2006). Although impacts are not limited to regions with poor services (e.g., treatment (Charron et al. 2004; Wade et al. 2014), the impact of floods on waterborne outbreaks will be modulated by the population density, underlying health status, and availability of health care (Watson et al. 2007). A better understanding of how floods can negatively affect health can also aid in prevention methods such as prophylaxis or vaccination campaigns against certain diseases that might increase in incidence after flooding (Dechet et al. 2012; Wu et al. 2008). Finally, future studies should pay special attention to any differential health effects that can arise from sporadic flooding compared to seasonal rains (e.g. monsoons) and associated flooding.

Climate regulation

Floods impact heterotrophic processes tied to the production and consumption of greenhouse gases (GHG: CO_2, CH_4, and to some extent N_2O) as a climate regulating ecosystem service provided naturally by soil systems. These processes include aerobic respiration of a wide range of organic compounds in floodwater (produces CO_2), methanogenesis (produces CH_4), and methane-oxidation (consumes CH_4). Other processes (e.g. acetate reduction) can produce CO_2 but are secondary in soil and will therefore not be discussed here. The primary process tied to N_2O production in soils is heterotrophic denitrification, or the reduction of NO_3^- into N_2 gas, which when incomplete leads to the production of N_2O gas (Naiman et al. 2005). Increased nitrogen supply during flooding may provide the raw materials for denitrification, but N_2O production is generally small in floodplains (Kaushal et al. 2014). Additionally, N_2O production following flooding is variable and relies on inundation time, substrate, and temperature (Kaushal et al. 2014; Pinay et al. 2000). A thorough review of the conditions (e.g., temperature, moisture availability, electron donors and acceptors) regulating these processes and associated GHG consumption or production can be found in Schlesinger and Bernhardt (2013). In addition to soil processes, flooding can transport large amounts of soil organic matter into aquatic ecosystems, where it can be processed further and release CO_2 (Richey et al. 2002).

Although translating changes in GHG fluxes at the soil-atmosphere interface into a single variable of air quality regulation remains a challenge, many studies have documented how GHG fluxes change in response to floods and water pulses at the soil-atmosphere interface (Kim et al. 2012). Although many more studies should be conducted to fully comprehend how GHG fluxes and associated air quality ecosystem services change following flooding events, some trends can be identified from published studies. In water limited environments where aerobic respiration is often limited by water availability, water additions / small floods generally lead to increased CO_2 emissions (Leon et al. 2014), but no consistent response across systems with respect to N_2O and CH_4. In a xeric environment (AZ, USA), Harms and Grimm (2012) show that following dry antecedent conditions, small floods typically stimulated CO_2 and CH_4 production, but not N_2O production. In wet and non-water limited environments, flood events typically lead to enhanced N_2O and CH_4 fluxes, especially under warm temperature conditions (> 20°C). Under wet antecedent conditions (monsoon season), muted CO_2 and N_2O responses were observed, while CH_4 emission increased following water additions (Harms and Grimm 2012). On the other hand, CO_2 fluxes under these conditions generally do not change drastically following storms as they mostly vary on a seasonal basis with higher CO_2 fluxes during summer months. In central New York state, USA, the remnants of Hurricane Irene and Tropical Storm Lee caused a large flood, which increased N_2O flux from 0.2 to 1.49 mg N/m^2/day and CH_4 flux from a range between -2 and 2 mg C/m^2/day pre-flood to 2.76 mg C/m^2/day post-flood, and increased short pulses in CO_2 at the onset of precipitation (Vidon et al. 2016a). In a water-limited forested riparian zone in North Carolina, USA, Vidon et al. (2016b) reported less negative CH_4 fluxes (i.e., methane oxidation decreased) and higher CO_2 fluxes (i.e., aerobic respiration increased) following water additions.

From an ecosystem services perspective, this suggests that if flood events become more frequent, ecosystems may present higher overall efflux of GHGs (Petrakis et al. 2017). Indeed, as indicated above, in water-limited environments, higher CO_2 production and associated emissions are likely to lead to overall increases in GHG emissions. In wetlands where strong CH_4 responses to storms are observed and where CH_4 can contribute large fractions of total GHG, an increased frequency in floods will also likely lead to overall increases in total GHG fluxes (e.g., Gomez et al. 2016). Finally, in hay and fertilized cornfields where CH_4 and N_2O combined can represent approximately 50% of total CO_2 emissions, floods are also likely to lead to overall increased GHG emissions (Bressler et al. 2017). It is only in non-water limited environments where most CO_{2eq} fluxes are generated by CO_2 emissions that floods are unlikely to have any significant impact on total GHG fluxes, as only muted CO_2 responses to storms are observed in these environments. Overall, climate and land use are therefore key factors to consider in assessing how floods might impact ecosystem services related to GHG induced changes in climate.

Provisioning services

Drinking water

Floods can impact drinking water when contaminants and pathogens are discharged into surface and underground drinking water sources. Any pollutants that are mobilized during flooding can impact drinking water sources. For example, flooding can increase total coliform (TC) concentrations by suspending sediment containing coliforms in rivers (Smith et al. 2008) or causing waste water from flooded sewer systems to infiltrate drinking supplies (Islam et al. 2007). Human wastes can also quickly infiltrate drinking water supplies during flooding in areas that lack proper waste disposal (Zahoor et al. 2016). Additionally, animal wastes can contaminate drinking water by contributing nutrients, pathogens, and metals (Burkholder et al. 2007). Metals stored in sediment can also be resuspended in aquatic ecosystems or enter drinking water sources through connectivity with contaminated water or runoff (Chrastny et al. 2006). Therefore, flooding has the potential to negatively impact drinking water supplies in a variety of ways.

For our literature survey, we considered a mixture of drinking water sources including drinking water reservoirs, wells, and taps. Here, we used TC and metal concentrations to assess the effects of flooding on drinking water. Limits on these parameters are among many criteria set for drinking water but are the most commonly reported in the literature. Nevertheless, TC and metal concentrations were only reported in the literature for extreme flooding. Therefore, we also included studies which quantified herbicides in drinking water supplies following flooding, including one study which quantified the herbicide atrazine after a small flood. These parameters were also included because they have significant health impacts when concentrations exceed drinking water standards. Bacteria present in drinking water can cause illnesses and even death in high-risk age groups such as children and the elderly (Figueras and Borrego 2010). Metal ingestion can have effects on the immune system, blood, liver, kidneys, and nervous system (Cempel and Nikel 2006).

In most studies, the quality of drinking water sourced from the tap or well water decreased after extreme flooding events. TC counts were compared to either local or more commonly World Health Organization (WHO) standards. Almost all well and tap water sampled after extreme flooding contained TC concentrations that exceeded drinking water standards (e.g. Chaturongkasumrit et al. 2013; Eccles et al. 2017; Islam et al. 2007). Metal concentrations measured included chromium, nickel, iron, lead, and cadmium. Most post-flood metal concentrations were elevated beyond pre-flood values in well and tap water (Zahoor et al. 2016) and in a drinking water reservoir (Chrastny et al. 2006). However, lead concentrations remained below World Health Organization (WHO) water quality standards after flooding in Lower Pakistan (Zahoor et al. 2016).

There were no results for the impact of small floods on either TC or metal concentrations. However, one study measured concentrations of the herbicide atrazine in drinking water sources following small floods. Small floods did not increase atrazine levels in drinking water supplies (Ray et al. 2002). Concentrations of the herbicides atrazine, alachlor, and cyanazine in well water also did not increase after extreme flooding (Chong et al. 1998). However, these results are influenced by the timing of herbicide application relative to the flood events. Flooding will likely mobilize recently applied herbicides from agricultural land and contaminate drinking water sources. One additional study which, used a water quality index found that drinking water quality decreased following seasonal flooding (Chen et al. 2015). Small floods can negatively impact drinking water, but there is a lack of evidence in this area to indicate the scope or prevalence of such impacts.

Food supply

Food sources that may be affected by flooding include fish, livestock, and crops. Flooding can increase soil regeneration and water availability for agriculture (Ogbodo 2011) or livestock and increase fish habitat and availability of food sources for fish (Jellyman et al. 2013). Small or seasonal flooding also is advantageous for native fish populations relative to invasive fishes occupying the same areas (Ho et al. 2013). However, extreme floods can destroy planted crops (Ferguson et al. 2012), drown livestock (Atta-ur-Rahman and Kahn 2013), and impair fish catch by reducing fish density (Endo et al. 2016). Fish production may increase or stay constant if an extreme flood falls within the normal flood regime that individual fishes are adapted to

(Lytle and Poff, 2004; Poff et al. 1997). Flood impact on fish populations is further complicated by flood timing. Floods that inundate large areas and occur when temperatures are warm are likely to result in hypoxia, affecting fish physiology, behavior, and survival (Pasco et al. 2015). Additionally, small floods that occur when temperatures are too low for native fish spawning may cause proliferation of invasive fish populations (Rayner et al. 2015). Communities which rely on subsistence farming and fishing are especially vulnerable to food reduction during and after flooding.

Most surveyed studies reported negative effects of extreme flooding on food supply. Several studies reported that crops were damaged during extreme flooding and that such flooding caused significant hardships for people who relied on farming as their main food source. Additionally, if extreme flooding extended into the next planting season farmers lost additional crops (Haile et al. 2013). Extreme flooding increased fish availability when floodwaters rose and receded. However, fewer fish were available when floodwaters were high (Sherman et al. 2015). In all studies, fish catch and consumption patterns were similar during small floods. People generally caught and consumed the least amount of fish during high water compared to periods of rising and receding floodwater (Isaac et al. 2015; Endo et al. 2016). Very few studies reported on flood impacts on livestock; however, one study reported that over 52,000 cattle drowned following an extreme flooding event that occurred in 2010 in Khyber Pakhtunkhwa, Pakistan (Atta-ur-Rahman and Kahn 2013). Therefore, extreme flooding negatively impacts food sources such as crops, fish, and livestock. There was an inadequate number of studies to determine the effects of small floods on agriculture. However, small floods should have either a net neutral or positive effect on agriculture due to increased water availability, more nutrients, and enhanced soil renewal processes (Ogbodo 2011). The importance of fish and crops as food sources differs depending on the society's location making it difficult to compare the relative importance of flooding. The effects of flooding on food supply also differ depending on the food source considered and at which stage of flooding food sources are quantified. For example, fish catch decreased during high water, but increased as water receded in a village on the banks of the Peruvian Amazon (Sherman et al. 2015). However, high water lasted months in some cases which was detrimental to people who rely on fish as a major part of their diets.

Cultural services

Aesthetic value

Aesthetic value refers to the view and natural qualities near water bodies that people find desirable. A flood, whether minor or major, can physically and functionally modify the ecosystem and infrastructure, which usually results in a reduction of the aesthetic value. Over longer term between extreme flood events, the aesthetic value generally recovers or can even be increased above the pre-flood value, depending upon the nature of the post-flood ecosystem recovery or shifts (e.g. Ronnback et al. 2007) and the implementation of post-flood management practices. Flood zone property values are generally enhanced by higher aesthetic value, but property values are also reduced by the perceived risk of floods (e.g. Shilling et al. 1985; MacDonald et al. 1987).

There was a lack of evidence for small floods affecting housing value, but extreme flooding led to decreased housing values in all cases. Home prices decreased markedly immediately following a flood event, particularly for lower priced properties in the 100-year flood plain, or in neighborhoods directly damaged by the flood (e.g. Bin and Polansky 2004; Eves and Wilkinson 2014). In contrast, higher priced properties in the 500-year flood plain were not found to decrease in value following a flood (Shultz and Fridgen 2001). This is attributed to a lack of awareness of home owners to the risks associated with the 500-year flood plain.

Recreation and tourism

Recreation refers to leisure activities that typically include fishing, boating, swimming, hunting, and hiking. Increases in river discharge can impact these activities by reducing safety with high flows and impaired water quality. However, higher water levels can also lead to enhanced fishing (Miranda and Meals 2013) and boating conditions (Stewart et al. 2003). The magnitude of flooding determines the effects on recreation. Major floods have a very immediate negative effect on recreation activities due to physical damage to infrastructure, ecosystems, and the loss of aesthetic value (Burger 2015). The long-term impact of a major flood on recreation is varied and depends strongly on the post-flood control and management of both information and recovery efforts. Tourism or ecotourism is related to recreation, but involves people traveling from outside the region, which generates additional economic value to nearby communities. Flooding may impact tourism by damaging infrastructure, reducing safety, damaging sites of interest, and changing tourist perceptions of an area (Walters et al. 2015).

From our literature review, we found that recreation is negatively impacted by extreme flooding. People were less likely to visit a recreational site, such as a park, after extreme flooding had occurred (Rung et al. 2011). Small floods had a general positive impact on recreation. Small experimental floods increased recreation by increasing the size and number of sandbars suitable for boats to stop at below the Glen Canyon Dam, Arizona (Stewart et al. 2003). Additionally, one study found that a study group comprised of students preferred rivers and streams located within parks to have dynamic hydrology (Eder and Arnberger 2016). Therefore, people are more likely to recreate in parks where natural water features have dynamic hydrology. Small floods increase hydrologic variability without causing the damages associated with extreme flooding. The effects of extreme flooding on tourism were mixed. Negative impacts included revenue losses (Kala 2014), evacuations (Faulkner and Vikulav 2001), and tourists deciding to avoid visiting the flooded area (Walters et al. 2015). These effects were temporary and tourism returned to pre-flood values after flood waters receded. In one study, tourists simply rescheduled their trips instead of traveling to an unaffected area (Faulkner and Vikulav 2001). It was also reported that flooded areas can appeal to travelers who want to help those affected (Walters et al. 2015). We were unable to make any conclusions on the impacts of small floods on tourism since we found no literature. However, there is some evidence that people tend to desire visiting areas with dynamic river systems so small floods may enhance tourism. As with recreation, the post-flood recovery efforts and the message communicated to the public play a crucial role (e.g. Walters et al. 2014). Education of the public through media presentations and outreach activities is very influential in restoring recreational activities. Having a disaster preparedness plan prior to an extreme flood, with effective implementation following a flood, can significantly improve the post-flood recovery in recreational and tourist activity (Faulkner and Vikulov 2001).

Conclusions

The influence of flooding on ecosystem services depends on flood size and service type with extreme floods more likely to be associated with declines in ecosystem services whereas small floods provide or enhance many ecosystem services (Figure 3.1; Table 3.3). Although we detected trends in ecosystem service availability following flooding, many services responded in complicated ways. Initial aquatic ecosystem conditions and time of year were important for determining whether a flood event, extreme or small, would result in gains or losses of a given ecosystem service. For example, floods occurring during warmer months with good light conditions were capable of causing algal blooms. However, a flood of the same size occurring in a different season may have no effect on primary production due to light limitation. Future research on the nuances involved with producing the ecosystem services addressed in this study should be done to improve our understanding of these services and how disturbances will affect them. Additionally, studies linking ecosystem processes with ecosystem services should be undertaken to improve our understanding of the effects of disturbance on aquatic ecosystem services in general.

River flooding is an essential component of natural flow regimes. However, against the backdrop of human-dominated systems, extreme floods were almost exclusively negatively associated with post-flood changes in aquatic ecosystem services (Table 3.3). More frequent extreme flooding will likely exacerbate losses in ecosystem services and possibly leave inadequate time for recovery between flood events. Ecosystem recovery following extreme floods is highly variable and can last months to years, depending on the effect considered (Swanson et al. 1998). For example, contaminant pulses resulting from extreme floods can be elevated for days to years post-flood (Kaushal et al. 2014). It is difficult to estimate ecosystem service recovery time following floods because monitoring typically does not extend beyond one post-flood measurement. Additionally, larger changes from pre- to post-flood could extend recovery time. Losses in ecosystem services such as drinking water and food supply will be especially detrimental in areas that lack drinking water filtration facilities (Delpla et al. 2009) and rely on subsistence farming (Haile et al. 2013) and fishing for food (Sherman et al. 2015). Approaches to reduce flood impacts on ecosystem services could include relocating agricultural land further from flood prone areas when possible, reducing impervious surfaces near water, reducing point and nonpoint pollution sources, and restoring riparian zones (Kaushal et al. 2014). However, there is much more work that needs to be done to find effective ways to manage extreme flooding.

Small floods were more likely to be associated with positive or neutral effects on ecosystem services (Table 3.3). However, small floods negatively affected water quality and disease regulation, but post-flood recovery may occur quickly because the magnitude of ecosystem service change following small floods is generally small compared to extreme floods. Additionally, these smaller floods typically occur seasonally and aquatic ecosystems are usually well-adapted to these disturbances (Junk et al. 1989). Many aquatic ecosystems do not experience these small beneficial floods because of damming and water regulating structures (Death et al. 2015), so there is no opportunity for flooding to enhance ecosystem service provisioning. Therefore, small floods should be favored as part of a healthy flow regime in aquatic ecosystems. Preserving natural flow variation that contributes to small floods is important for aquatic ecosystems and as shown here, ecosystem service provision. Activities which preserve the occurrence of small floods include decreasing impervious surfaces and restoring riparian areas to reduce runoff that increases flood magnitude (Ogden et al. 2011) and limiting the extent of flow alteration such as refraining from building dams (Acreman et al. 2014).

Many previous studies have reported that dynamic flow regimes that include floods, even occasional extreme floods, are ecologically important (Peters et al. 2016) but few have linked floods with aquatic ecosystem service provisioning. We evaluated ecosystem service gains and losses in response to flooding and identified possible mechanisms that lead to these changes (Figure 3.3) and found that aquatic ecosystems require flood protection strategies designed to dampen the undesired effects of extreme floods and enhance smaller beneficial floods to maximize ecosystem service provision. There are many methods available to do this including restoring lateral connectivity between the river and floodplain, regenerating functional riparian areas (Death et al., 2015), reconnecting fragmented aquatic ecosystems to reduce runoff, and reforesting headwaters (Barbedo et al. 2014). Not all floods can or should be prevented, but these strategies in combination should improve flood regulation without exerting the negative impacts commonly associated with flood mitigation practices. However, we must be diligent in designing and implementing these plans as quickly as possible because current and future increases in flood magnitude will be deleterious to aquatic ecosystems and reduce aquatic ecosystem services. Ecosystem services examined in this study represent some of the essential life sustaining benefits that people gain from aquatic ecosystems such as food supply, drinking water, and human disease regulation. Flood protection strategies that are effective at reducing the damages caused by extreme flooding will have profound benefits beyond protecting our built infrastructure. They will also protect the aquatic ecosystems and their ecosystem services that we rely on for health and survival.

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Table 3.1. Ecosystem services with indicators used to capture ecosystem service changes, indicator units, process linking ecosystem service with flooding, and ecosystem service type as defined in the Millennium Ecosystem Assessment.

Ecosystem service	**Indicator**	**Unit**	**Process**	**Type**
Primary production	NPP, GPP	mg C/m^3/time	Changes in nutrients and physical conditions impact NPP/GPP	Supporting
Soil formation	Erosion, accumulation volume	m^3	Sediment deposition on shores/ more sediment transport in water	Supporting
Water regulation	Groundwater and aquifer volume or height	m^3, m	Water retained in ecosystem for some anthropogenic use (drinking, irrigation, etc)	Regulating
Water quality	Water nitrogen and phosphorus concentration	µg/L, mg/L	Increased nutrient transport	Regulating
Regulation of human disease	Odds ratio	none	Release of disease-causing agents from sediment or overflowing sewer systems	Regulating
Climate regulation	Methane and carbon dioxide release	g CH_4/time	Changes in aerobic/anaerobic microbial processes that influence organic matter decomposition	Regulating

Drinking water	Total coliform, metal concentrations	cfu/ml, mg/L	Bacteria and metals mobilized by floodwaters and enter drinking water sources	Provisioning
Food supply	Crops damaged, change in fish catch	none	Crops destroyed by physical impacts of floodwater, changes in fish distribution and abundance	Provisioning
Aesthetic value	Housing value discount	$	Damage and risk of flooding reduce desire to live near water	Cultural
Recreation and tourism	Willingness to visit recreation area, revenue lost	$	Algal bloom, unsafe water levels, debris in water, lack of infrastructure to travel to destination	Cultural

Table 3.2. Examples of quantitative changes in climate regulation and disease regulation ecosystem service indicators, where pre-flood, post-small flood, and post-extreme flood values were derived from the same study.

Ecosystem service	Location	Indicator	Pre-flood value	Post-small flood	Post-extreme flood	Reference
Climate regulation	Danube River, Austria	CH_4 flux (µmol/m^2/h)	72.2	77.4	303.2	Sieczko et al. 2016
Regulation of human disease	China	Odds ratio	1.00	1.14	1.28	Gao et al. 2016

Table 3.3. Summary of the impacts of small and extreme floods on ecosystem service gains and losses. Gains are expressed as “+”, losses as “-“, and neutral effects as “0”.

Ecosystem service	**Gains or losses (+/-/0)**	
	Small flood	**Extreme flood**
primary production	+	+
soil formation	-	-
water regulation	+	+
water quality	-	-
regulation of human disease	-	-
climate regulation	0	-
drinking water	0	-
food supply	-	-
aesthetic value	NA	-
recreation and tourism	+	-

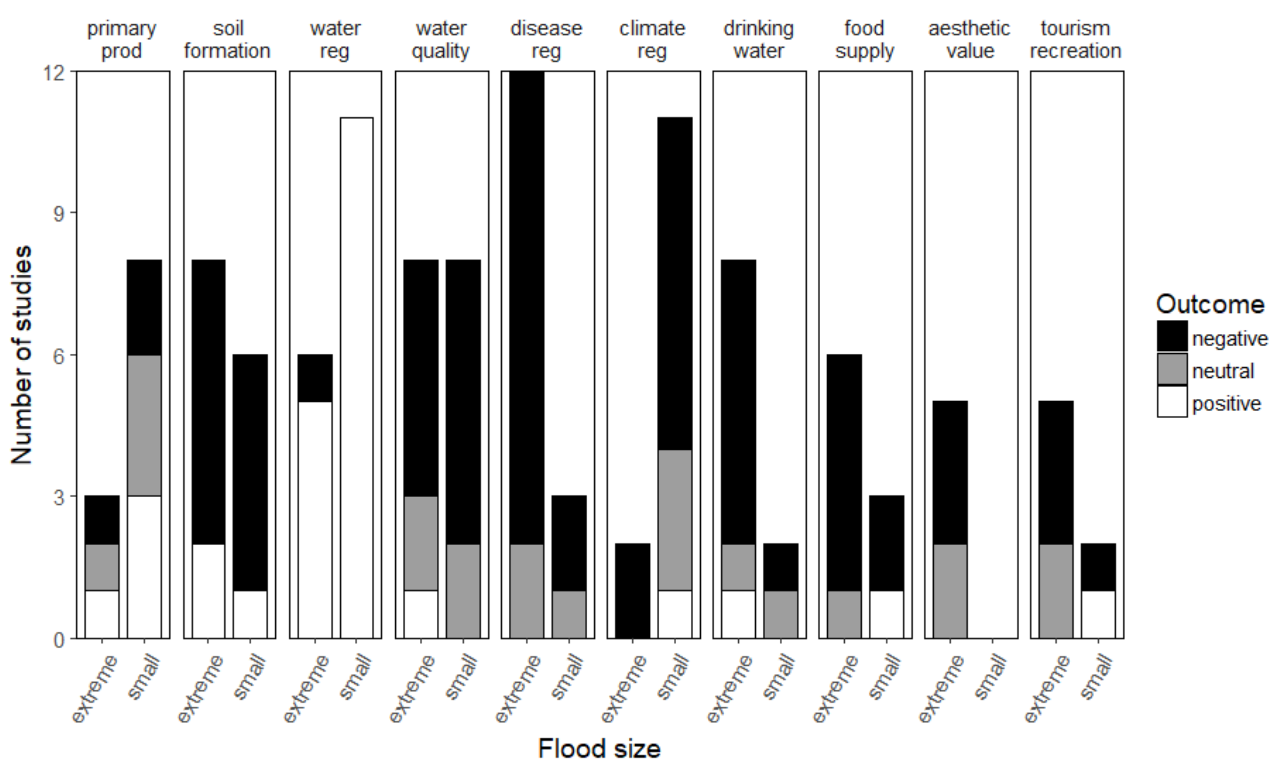

Figure 3.1. Number of studies resulting from a systematic literature review with negative, neutral, and positive outcomes on ten aquatic ecosystem services following small and extreme floods

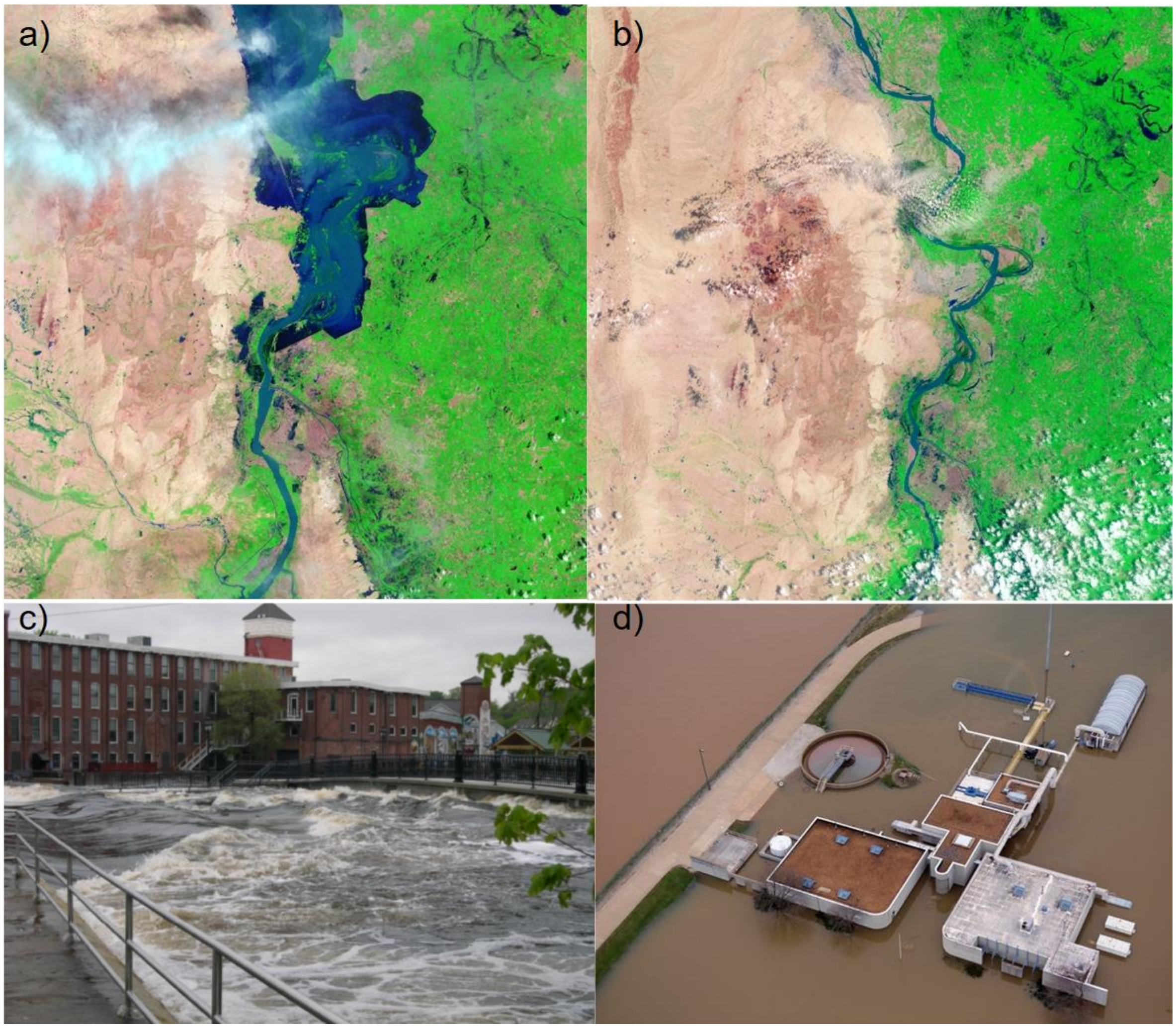

Figure 3.2. Photos of flooding taken from different perspectives. Satellite photos of extreme flooding (a) and seasonal flooding (b) on the Indus River, Pakistan, ground level photo of extreme flooding on the Ipswich River, Massachusetts, USA (c) and aerial photo of extreme flooding engulfing a sewage treatment plant on the Meramec River, Missouri, USA (d). Image sources: NASA Earth Observatory, https://earthobservatory.nasa.gov/IOTD/view.php?id=45393 (a-b), Wilfred Wollheim (c), David Carson, St Louis Post-Dispatch (d)

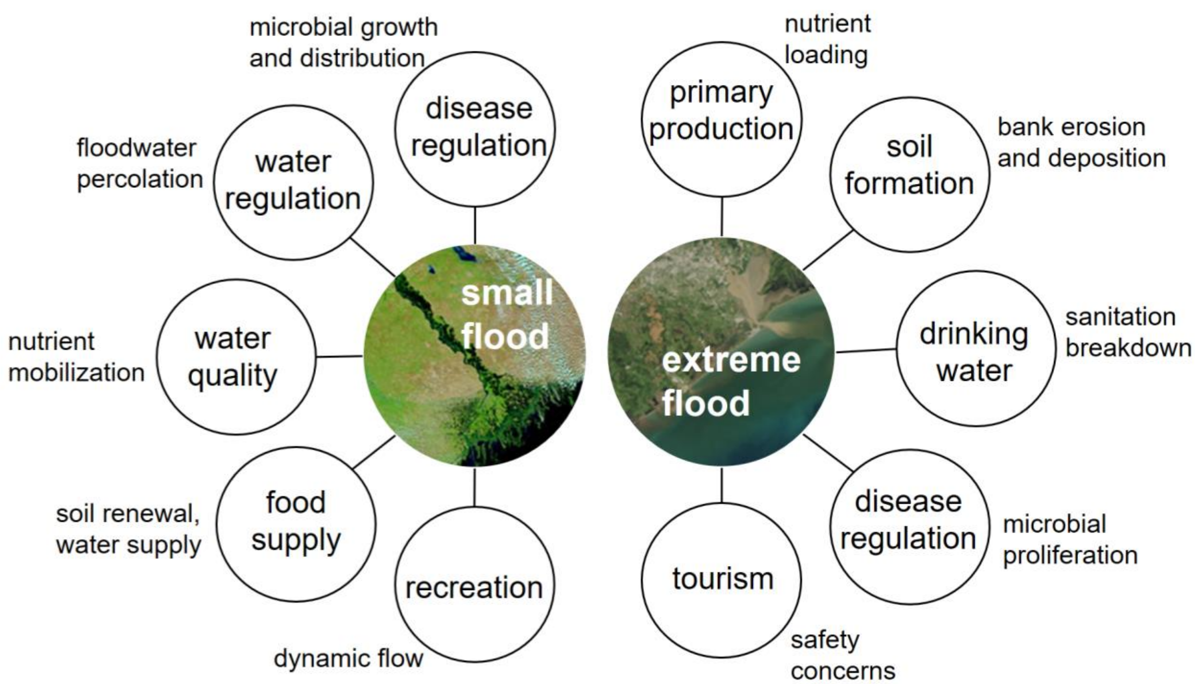

Figure 3.3. Processes linking small and extreme floods to changes in aquatic ecosystem services. Image sources: NASA Earth Observatory, https://earthobservatory.nasa.gov/NaturalHazards/view.php?id=14932&eocn=image&eoci=related_image (left) and https://earthobservatory.nasa.gov/IOTD/view.php?id=90703 (right).

4. General Conclusions

In this study, we evaluated the effects of flooding using a combination of quantitative and qualitative approaches. Flooding exerts complex effects on aquatic ecosystems and we found that there is value in assessing these effects in different ways. We also considered two different flooding locations: terrestrial areas unaccustomed to flooding and floodplains adjacent to rivers. The current state of knowledge regarding the effects of flooding is lacking in many areas and the relatively unpredictable nature of flood events makes capturing pre- and post-flood changes difficult. However, we were able to make some conclusions about how changes in flooding will affect aquatic ecosystems and ecosystem services.

We explored the unique dynamics of N and P during flooding and quantified their release from organic matter using data from a whole-ecosystem flooding experiment. The results of this analysis are important for understanding the effects of increases in the flooding of organic matter from changes in precipitation and from artificial flooding for reservoir creation. We found that flooding organic matter caused substantial amounts of N and P to be released. The magnitude of N and P release, however, was not related to the site carbon content, suggesting that the type of vegetation flooded may have a larger influence. More N than P was released throughout the experiment, increasing the N:P molar ratio of reservoir outflows relative to inflows. Consequently, flooding organic matter naturally or artificially (i.e. reservoir creation) will contribute to nutrient loading in receiving systems. It is unlikely that we will be able to reduce nutrients contributed by flooded organic matter, especially in natural areas. However, decreasing the number of reservoirs built in areas with high amounts of stored organic matter or removing organic matter prior to flooding may reduce the quantity of nutrients released. Additionally, it would be beneficial to assess aquatic ecosystems downstream of new reservoirs to gain a better understanding of how post-flood nutrient enrichment will affect that system.

The effects of flooding are complex and studies measuring changes in biophysical data are intrinsically important, but these data are also valuable for assessing how flooding will affect society. To further explore the effects of flooding, we completed a qualitative analysis of the effects of flooding on aquatic ecosystem services. We were able to identify gains and losses in 10 aquatic ecosystem services after the occurrence of small (<10 year return interval) and extreme floods (> 100 year return interval) using a systematic literature review approach. Small floods tended to cause gains in ecosystem services, while extreme floods generally had a negative influence on ecosystem services. However, the list of aquatic ecosystem services considered is not exhaustive and the duration of the ecosystem service losses or gains was difficult to assess. Our evaluation of aquatic ecosystem services provided insight into the importance of flooding for maintaining and enhancing some services, while beginning to help us understand how increases in extreme floods may reduce ecosystem service provisioning in the future. Flow regimes that include floods will be especially beneficial for aquatic ecosystem services. However, more frequent extreme floods may reduce ecosystem service provisioning. Through this analysis, we also obtained a better understanding of the current state of flood-related research as it relates to people and where future research efforts are needed. Generally, better quantification of biophysical changes pre- and post-flood will help us to better understand the dynamics of how flooding will impact ecosystem services.

Appendix A

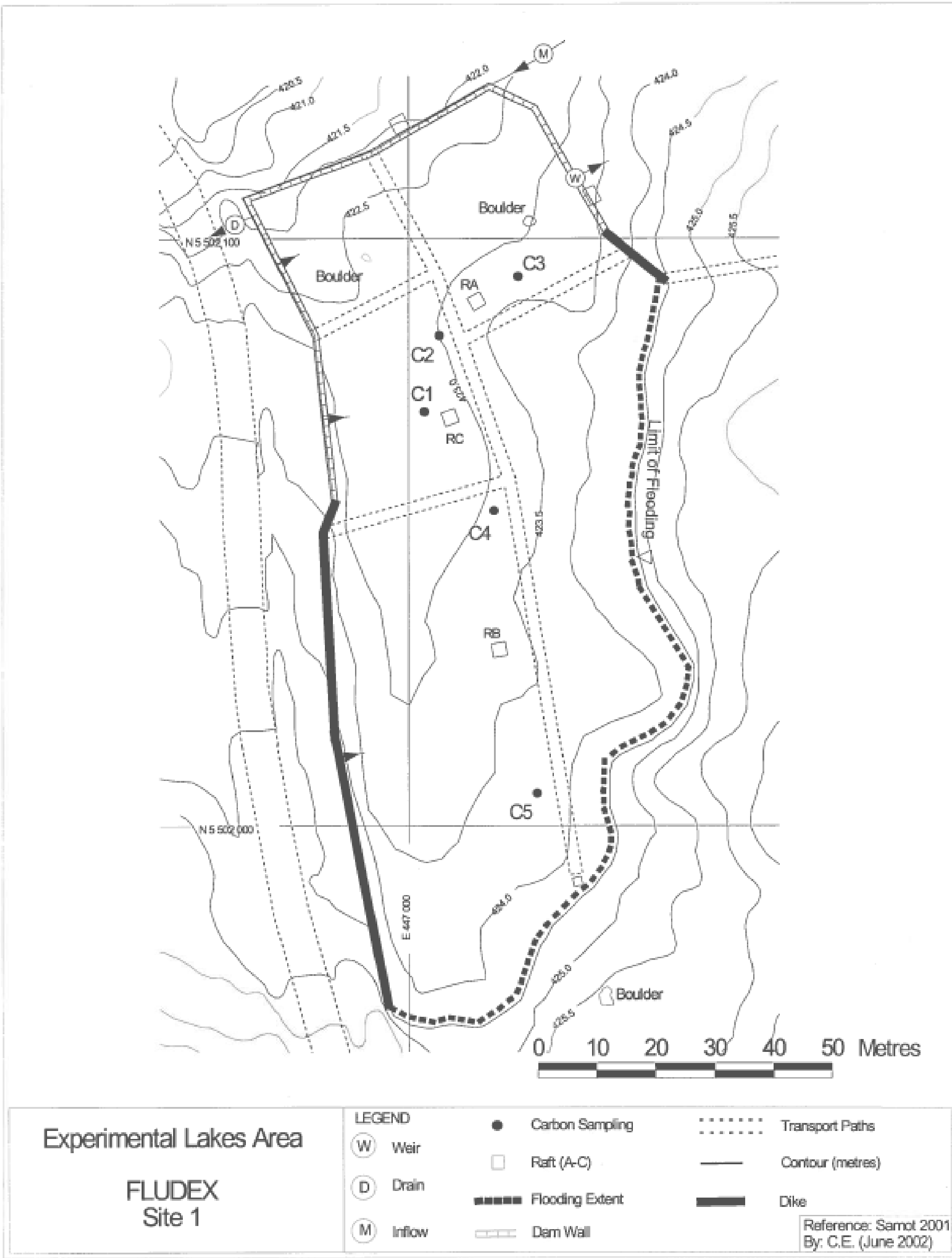

Figure A1. Topographic map of FLUDEX site 1 (high carbon) at the Experimental Lakes Area (ELA) in Northwestern Ontario.

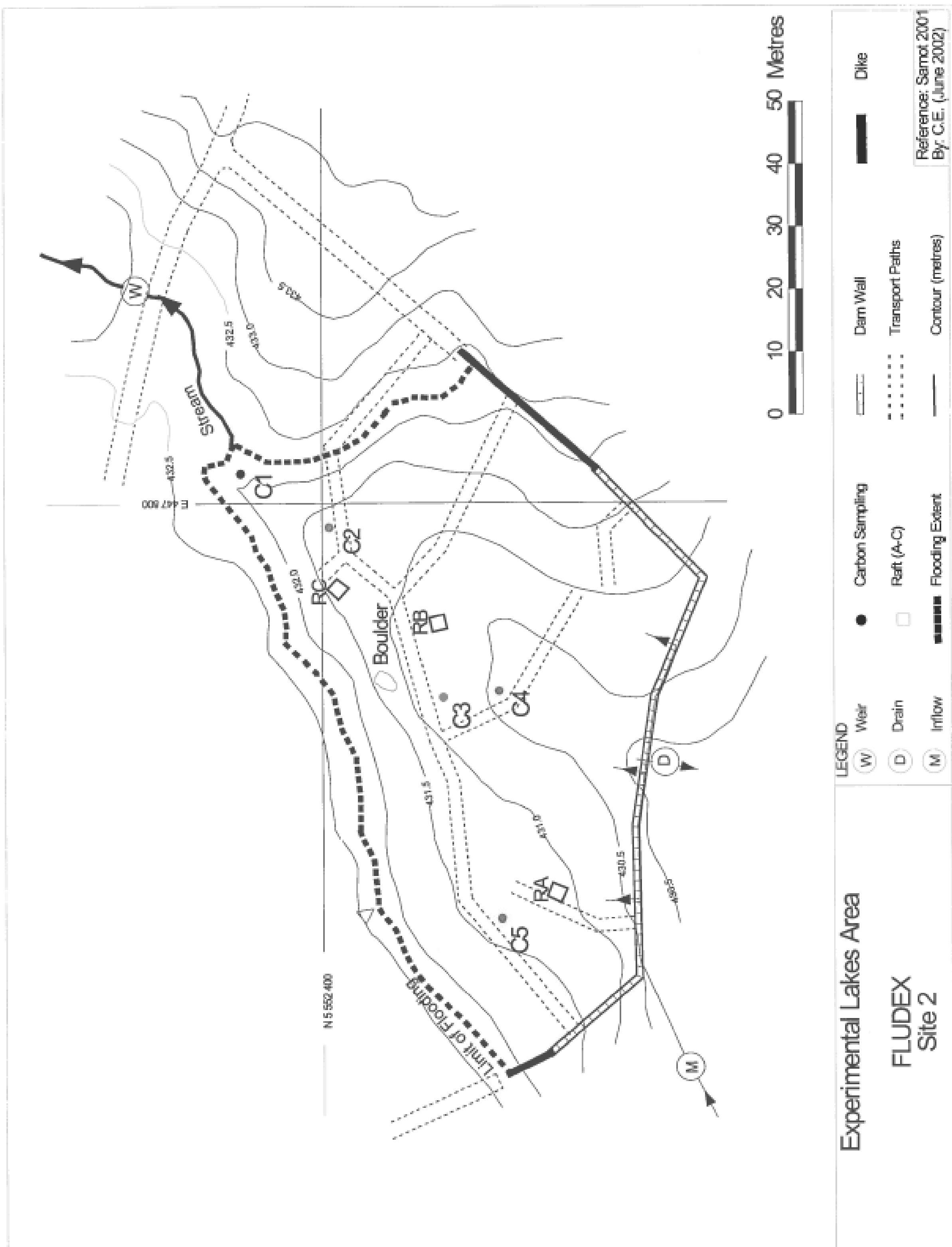

Figure A2. Topographic map of FLUDEX site 2 (medium carbon) at the Experimental Lakes Area (ELA) in Northwestern Ontario.

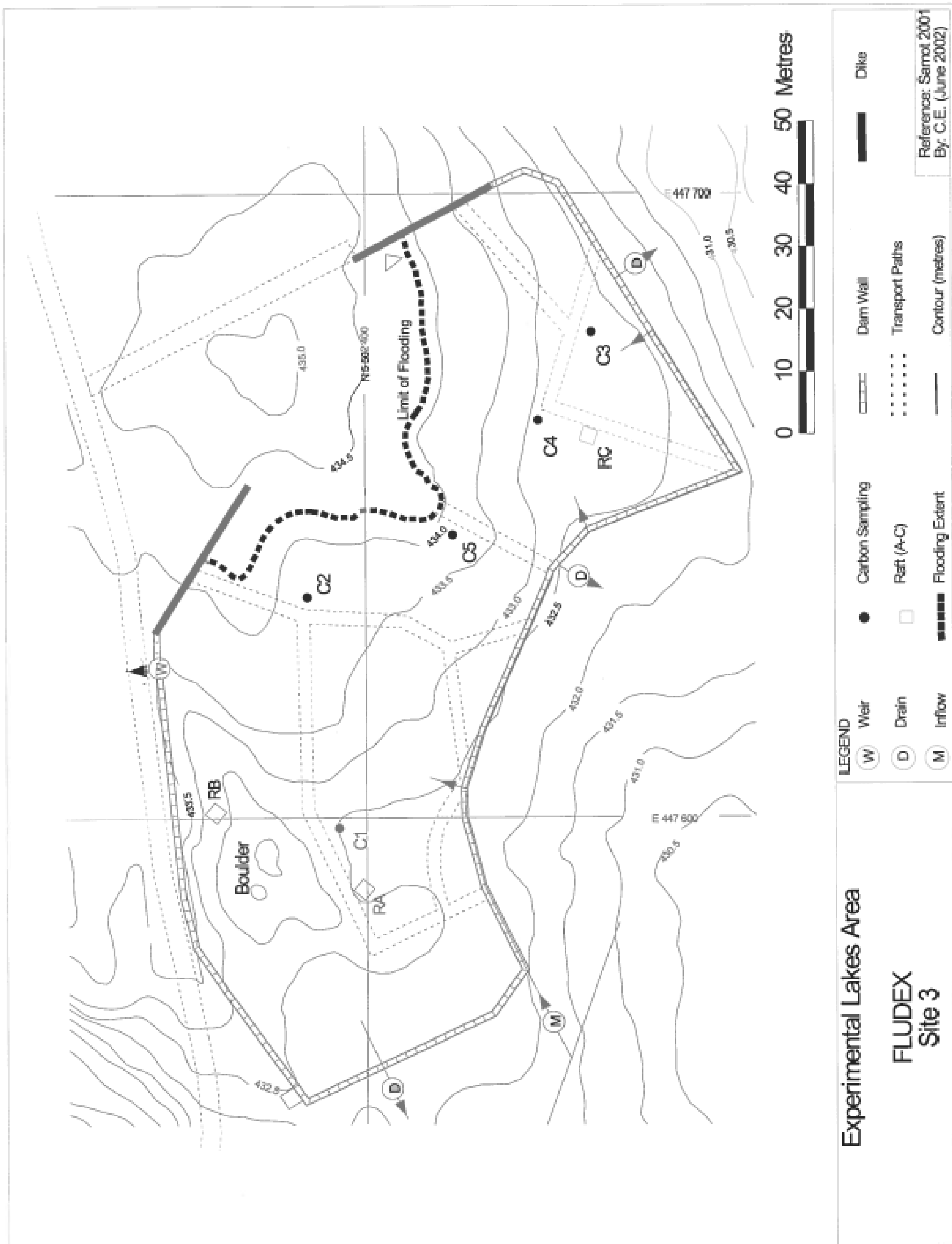

Figure A3. Topographic map of FLUDEX site 3 (low carbon) at the Experimental Lakes Area (ELA) in Northwestern Ontario.

Appendix B

Table B1. TP masses in each component of the low, medium, and high carbon reservoir TP mass budgets in g ha^{-1}.

	Low carbon					Medium carbon					High carbon				
Year	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003
Inputs															
Inflow	1209	1382	772	765	1204	727	1045	891	603	728	530	986	679	640	772
Precipitation	110	482	313	139	100	110	482	313	139	100	110	482	333	139	100
Runoff	4	7	3	3	0	39	67	31	28	4	174	294	139	122	16
Total	1323	1870	1088	907	1304	876	1593	1235	769	832	813	1762	1151	901	888
Outputs															
Weir	1354	1120	1188	660	648	4013	2265	2025	1348	1181	2096	1827	1468	1826	1180
Drain	111	100	89	100	111	111	102	119	119	94	138	110	138	147	156
Seepage	2447	1623	1160	881	1005	909	453	269	129	232	583	428	441	482	458
Total	3911	2843	2436	1640	1763	5033	2820	2413	1597	1507	2817	2365	2047	2455	1794
Flux	2588	972	1349	733	459	4157	1227	1178	827	675	2003	603	896	1554	906

Table B2. TN masses in each component of the low, medium, and high carbon reservoir TN mass budgets in g ha^{-1}.

	Low carbon					Medium carbon					High carbon				
Year	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003
Inputs															
Inflow	46599	44343	33231	37656	60695	44290	33122	40499	30119	37161	32323	32573	31183	31926	39520
Precipitation	3135	5524	4776	5554	3589	3135	5524	4776	5554	3589	3135	5524	5034	5554	3589
Runoff	94	229	112	125	15	934	2278	1114	1244	152	4118	10040	4912	5483	668
Total	49828	50096	38119	43335	64299	48360	40924	46390	36916	40902	39576	48137	41130	42962	43777
Outputs															
Weir	22700	23164	29659	20964	29662	59713	45779	53644	39724	39178	36825	41570	33947	42502	43334
Drain	4074	3100	3742	3565	4739	3353	2696	3430	3191	3780	3557	3658	3566	3750	4476
Seepage	39792	33549	28945	28109	43218	12964	9020	7057	3775	7480	10021	9792	10178	11092	16066

Total	66566	59813	62346	52638	77618	76030	57495	64131	46690	50438	50403	55020	47692	57343	63876
Flux	16738	9716	24227	9302	13319	27671	16571	17741	9774	9537	10827	6883	6562	14381	20099

Appendix C

NH_4 was the dominant inorganic N species in all reservoirs from 1999 to 2003. NH_4 concentrations were especially high in the bottom of the medium carbon reservoir (939 µg L^{-1}) in 1999 (Figure C1). After 1999, NH_4 was generally similar in the low and medium carbon reservoirs. NH_4 concentrations were generally highest in the high carbon reservoir in 2000 and 2002, but similar among all reservoirs in 2001 and 2003. During the first year of flooding in 1999, NO_2 concentrations were also high in all three reservoirs. In the low and medium carbon reservoirs, NO_2 concentrations decreased to below detection and remained that way for most of 2000 and 2002 (Figure C2). In the high carbon reservoir, NO_2 concentrations were slightly elevated in the bottom (4 µg L^{-1}), middle (1.3 µg L^{-1}), and surface (1 µg L^{-1}) of the reservoir in the first two weeks of flooding in 2000. NO_2 concentrations were below detection throughout 2002 but increased again in the beginning of 2003 (2 µg L^{-1}) and were again below detection at the end of 2003. NO_3 concentrations were below detection in all reservoirs during the first year of flooding in 1999. In 2000 to 2001, NO_3 concentrations increased in all reservoirs but were highest in the medium and low carbon reservoirs (Figure C3). In 2002, NO_3 concentrations were still elevated and were similar in all three reservoirs. In 2003, NO_3 concentrations in all reservoirs were below detection for the first ten weeks of flooding and then increased to 4 µg L^{-1} in the low and medium carbon reservoirs and 6 µg L^{-1} in the high carbon reservoir.

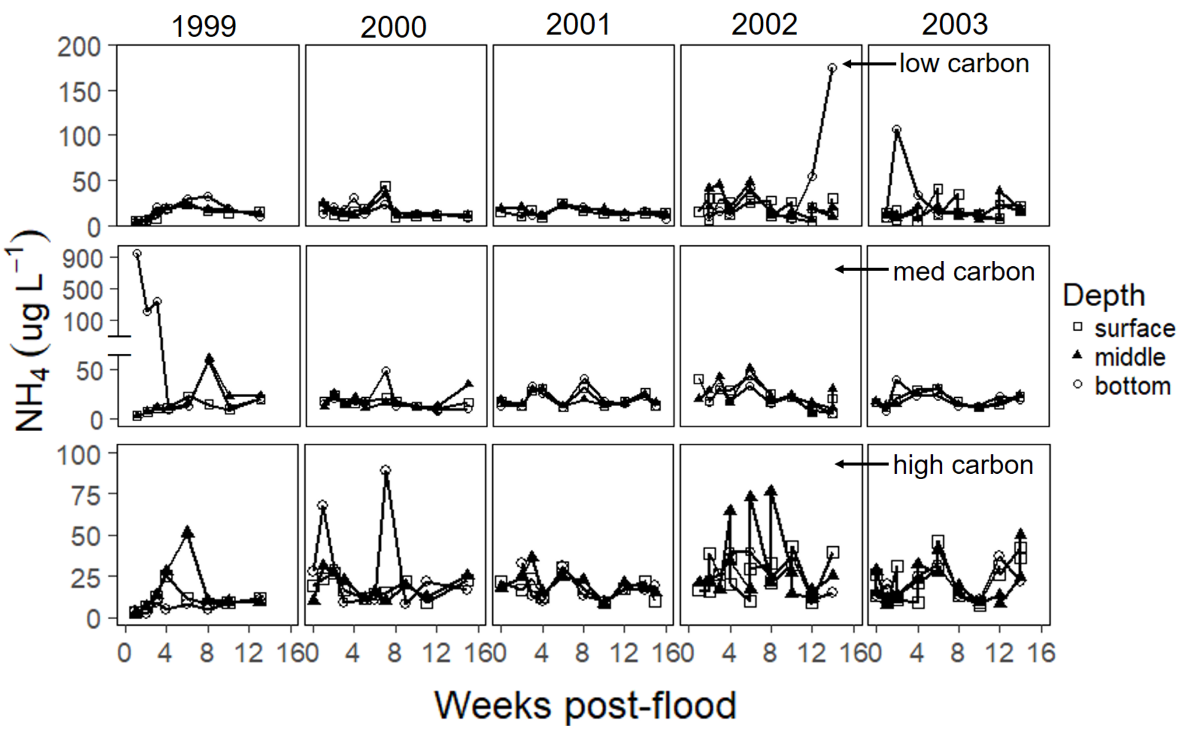

Figure C1. NH_4 concentrations measured at three different depths in the low (top row), medium (middle row), and high (bottom row) reservoirs from 1999 to 2003.

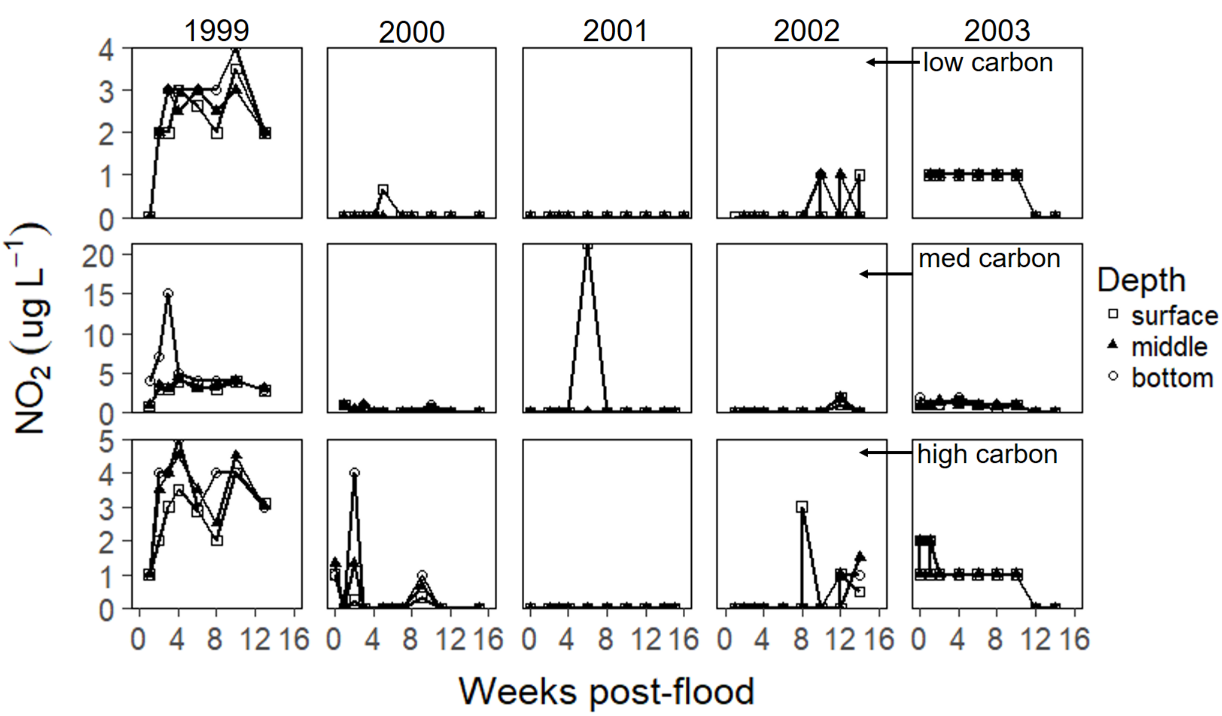

Figure C2. NO_2 concentrations measured at three different depths in the low (top row), medium (middle row), and high (bottom row) reservoirs from 1999 to 2003.

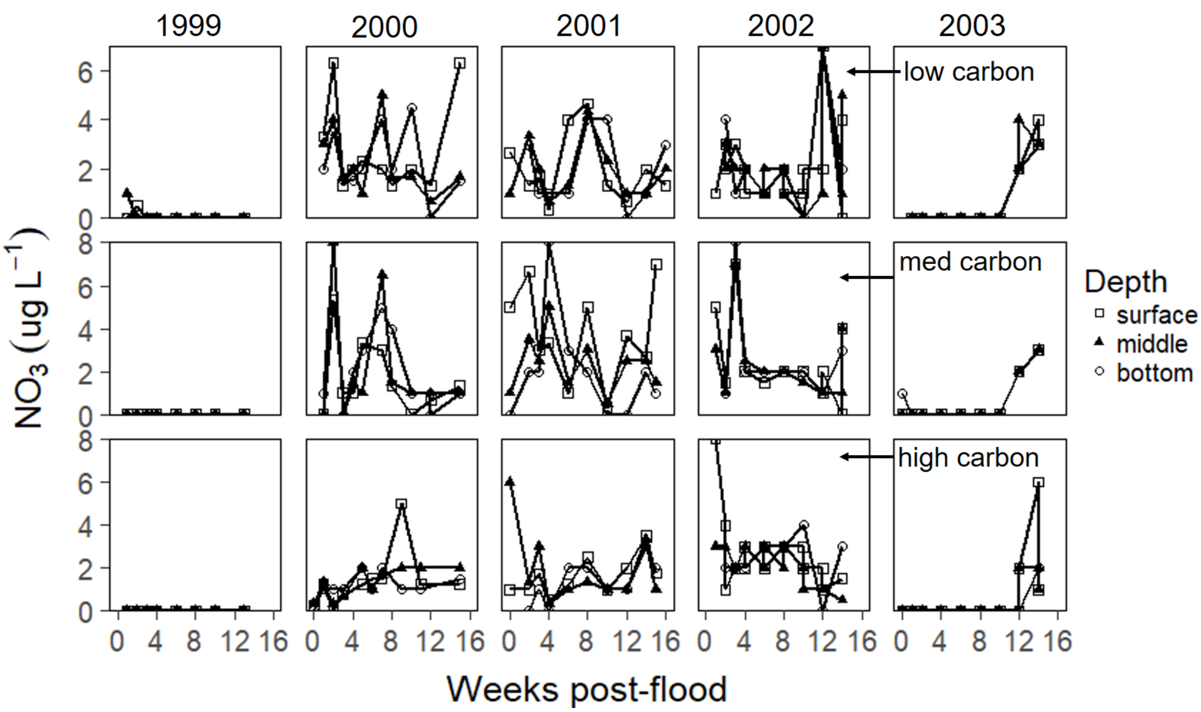

Figure C3. NO_3 concentrations measured at three different depths in the low (top row), medium (middle row), and high (bottom row) reservoirs from 1999 to 2003.

Appendix D

Table D1. Number of sources found and country where each study was completed for each ecosystem service and flood size.

Service	Flood size	Number of papers	Country (no. papers)
Primary production	small	8	Australia (1), Botswana (1), Brazil (1), Peru (1), Spain (1), Switzerland (2), USA (2), Venezuela (1)
	extreme	3	Botswana (1), Brazil (1), USA (1)
Soil formation	small	6	Australia (1), Bangladesh (1), Cambodia (1), India (1), Pakistan (1), USA (1)
	extreme	8	Bangladesh (1), Canada (1), France (2), New Zealand (1), Pakistan (2), USA (1)
Water regulation	small	11	Australia (1), Cambodia (1), China (1), Ethiopia (1), Mali (1), Namibia (3), Niger (1), Spain (1), Turkey (1), USA (1)
	extreme	6	Australia (1), Chile (1), China (1), Namibia (1), Spain (1), USA (1)
Water quality	small	8	Brazil (1), China (1), Croatia (1), France (1), Italy (1), Morocco (1), USA (2)
	extreme	8	Austria (1), Germany (1), Poland (1), Taiwan (1), USA (4)
Regulation of human disease	small	3	Cambodia (1), China (1), Sudan (1)
	extreme	12	Bangladesh (2), Cambodia (1), Canada (1), China (3), Germany (1), Sudan (1), Uganda (1), USA (2)
Climate regulation	small	11	Austria (1), Botswana (1), Brazil (4), Spain (1), USA (3), Venezuela (1)
	extreme	2	Botswana (1), USA (1)
Drinking water	small	2	China (1), USA (1)
	extreme	8	Australia (1), Bangladesh (1), Canada (1), Czech Republic (1), Germany (1), Pakistan (1), Thailand (1), USA (1)
Food supply	small	3	Brazil (2), Pakistan (1)
	extreme	6	Bangladesh (1), Ethiopia (1), India (1), Mozambique (1), Pakistan (1), Peru (1)
Aesthetic value	small	none	None
	extreme	5	Australia (1), USA (4)
Recreation and tourism	small	2	Germany (1), USA (1)
	extreme	4	Australia (2), India (1), USA (1)