Key components and contrasts in the nitrogen budget across a US-Canadian transboundary watershed

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Key Points:

- Nearly 81% of nitrogen inputs to the Nooksack River Watershed were used to support agricultural production, most of which was animal feed
- The largest export was in the form of ammonia from the agriculture sector (32%)
- Different policy frameworks between US and Canada had impacts on components on nutrient management in different portions of the watershed

Keywords:

Nitrogen budget, transboundary watershed, ammonia emission, nitrogen use efficiency, agriculture, land use

Abstract

Watershed nitrogen (N) budgets provide insights into drivers and solutions for groundwater and surface water N contamination. We constructed a comprehensive N budget for the transboundary Nooksack River Watershed (BC, Canada and WA, US) using locally-derived data, national statistics and standard parameters. Feed imports for dairy (mainly in the US) and poultry (mainly in Canada) accounted for 30 and 29% of the total N input to the watershed, respectively. Synthetic fertilizer was the next largest source contributing 21% of inputs. Food imports for humans and pets together accounted for 9% of total inputs, slightly lower than atmospheric deposition (10%). Returning salmon represented <0.06% of total N input but was an important ecological flux by importing marine-derived nutrients. Quantified N export was 80% of total N input, driven by ammonia emission (32% of exports). Animal product export was the second largest output of N (31%) as milk and cattle in the US and poultry products in Canada. Riverine export of N was estimated 28% of total N export. The commonly used crop nitrogen use efficiency (crop NUE) alone did not provide sufficient information on farming activities and should be combined with other criteria such as farm-gate NUE to understand management efficiency. Agriculture was the primary driver of N inputs to the environment despite widespread adoption of conservation practices, illustrating a need to optimize management to minimize hydrologic and volatilization losses. The N budget provides key information for stakeholders across sectors and borders to create environmentally and economically viable and effective solutions.

1 Introduction

The production and consumption of food and energy are increasing the cycling of reactive nitrogen in the environment (Davidson et al., 2011; Galloway et al., 2004; van Meter et al., 2016). While the usage of N to produce food and energy sustains human health and well-being, intentional and unintentional release of excess N has led to significant ecological consequences, such as eutrophication of fresh and coastal waters, hypoxia of aquatic systems, contamination of drinking water, degradation of air quality, deposition-induced acidification, and loss of biodiversity (Baron et al., 2011; Greaver et al., 2012; Pennino et al., 2017). Developing the best available information on N sources and transport is needed at different scales to promote effective management activities, yet this is a challenging task because of the wide variety of sources, forms, processing and loss vectors along the N cascade (Alexander et al., 2009; Erisman et al., 2003; Galloway et al., 2003).

One useful approach to bridge the gap between N flows and nutrient reduction goals can be found by assembling integrated, multi-source, multi-sectoral N budgets for specific areas of concern. The creation of a N budget is an essential step towards an integrated approach to solving problems associated with N release. Input-output budgets can help decision-makers better understand and manage N release by providing quantification of N fluxes at scales appropriate for making management decisions. Many types of accounting approaches have emerged to provide decision-makers information about N sources and loadings (such as NANI, SPARROW, WSAM) (Hong et al., 2011; Sprague et al., 2000; Swaney et al., 2018). These efforts have provided information at county, state and country scales, but N in the environment does not follow geopolitical boundaries. Through long-range transport in the atmosphere and waters, the environmental impacts of N can extend from local to regional to continental to global scales, depending on the form and fate (Erisman et al., 2003; Galloway, 2003). Partnerships between countries and institutions may assist in development and implementation of effective N management, especially where N crosses international boundaries. Successful partnership examples on other environmental issues include the Great Lakes Water Quality Agreement between the United States (US) and Canada that works to develop new nutrient reduction targets and explore pathways to reach the common goal (Team, 2015), and the Baltic Sea Action Plan, a multinational collaboration that has made great progress in reducing nutrient inputs to the Baltic (McCrackin et al., 2018).

Straddling the border of Washington State, US, and British Columbia, Canada, the Nooksack River Watershed (NRW) supports agriculture, fisheries, wildlife, and urban communities from the North Cascades to Bellingham Bay in Puget Sound, and from the Fraser River towards Vancouver, BC. Agricultural land in the watershed is dominated by forage crop production supporting confined animal operations (dairy and poultry) as well as berries. Land application of livestock manure is a common agricultural practice as a source of nutrients for crop production (Bittman et al., 2019; Cox et al., 2018). Excess N in both air and water have elevated both environmental and human health risks in the watershed. Caused by enhanced N emission to the atmosphere and subsequent deposition, exceedances of N critical loads were observed or expected in urban and agricultural corridors in this region, which can potentially lead to significant harmful effects on local species and a cascade of effects on the entire ecosystem (Baron et al., 2011; Geiser et al., 2010; Greaver et al., 2012; Sheibley et al., 2014). Elevated N emission can impair air quality by lowering visibility and contributing to particulate matter and ozone precursors that are harmful to human health.

For decades, groundwater nitrate concentrations have exceeded the maximum contaminant level (MCL) for drinking water (10 mg L^{-1}) in the transboundary Sumas-Blaine Aquifer (SBA) (Zebarth et al., 2015). The SBA, which partially overlaps the NRW, is the primary source of drinking water for the transboundary area (Carey et al., 2017) (Figure 1). About 29% of private wells sampled on the US side of the SBA exceeded the MCL (Carey & Cummings 2013). Recent studies have shown decreasing trends both in nitrate concentrations in some wells and in the total number of monitoring wells exceeding the MCL (Carey et al., 2017), but high nitrate concentrations in drinking water wells remain a concern in the area (Cox et al., 2018).

The transboundary nature of the watershed has complicated efforts to trace N pollutant sources in air and waters and to develop effective nutrient management plans. Construction of a transboundary total N budget allows us to integrate information from all sectors, compare different management practices across the border, and link these activities to the environmental outcomes. An informal partnership formed in 2016 between scientists and stakeholders in the US and Canada to study N budgets and sustainability for the transboundary watershed. The NRW N budget project is the North American demonstration for the International Nitrogen Management System (INMS), which aspires to bring together scientists and communities to improve nitrogen management across the globe (http://www.inms.international/about_INMS).

The objectives of the NRW budget study were: 1) to construct the first comprehensive N -budget of the NRW using local data on N sources and exports; and 2) to combine the information on cross-boundary N inputs and outputs to gain a better understanding of local N retention and transport mechanisms and N use efficiencies. We hope to use the binational N budget findings to facilitate future studies on how differences in management and policies affect

N fates in the environment, which could help create environmentally effective and economically viable solutions to improve air and water qualities in the region.

2 Study area

The headwaters of the Nooksack River are in the western North Cascade Mountains (Mt. Baker and Mt. Shuksan), and the river flows west through lowlands before discharging to Bellingham Bay north of the city of Bellingham. The Nooksack River drains an approximately 2130 km^2 area of northwest Washington State in the US and southwestern British Columbia in Canada. Most of the watershed area is in the US (94%; Figure 1). Mean annual discharge ranges from 80 to 110 cms (Dickerson-Langer & Mitchell, 2014). The watershed climate is a mixture of temperate maritime and Mediterranean-type according to the Köppen climate classification (Kottek et al., 2006). About 70% of annual rainfall occurs from October to March (Cox et al., 2018), and summers (July-October) are generally dry (Pelto 2015). About 80% of the watershed area lies in mountainous forests dominated by coniferous trees. Urban and residential land together is about 10% of the watershed area, with a total population of over 110,000 people. The agricultural land area in 2014 was about 174 km^2 on the US side and 42 km^2 on the Canadian side, comprising 10% of the land area of the watershed. In 2014, cultivation of forage crops (grass and corn) together accounted for about 63% of agricultural land on the US side (WSDA, 2015), while berry crops dominated on the Canadian side, accounting for 80% of cropland. Much of the crop production on the US side supports dairy operations, which remain an important economic component in the state despite recent declines in the state's animal populations (Cox et al., 2018; USDA, 2017). In 2014, there were over 30,000 dairy cows on the US side of the watershed. On the Canadian side, poultry farms were the major animal production with a 2014 accumulated chicken population in the watershed of over 127 million.

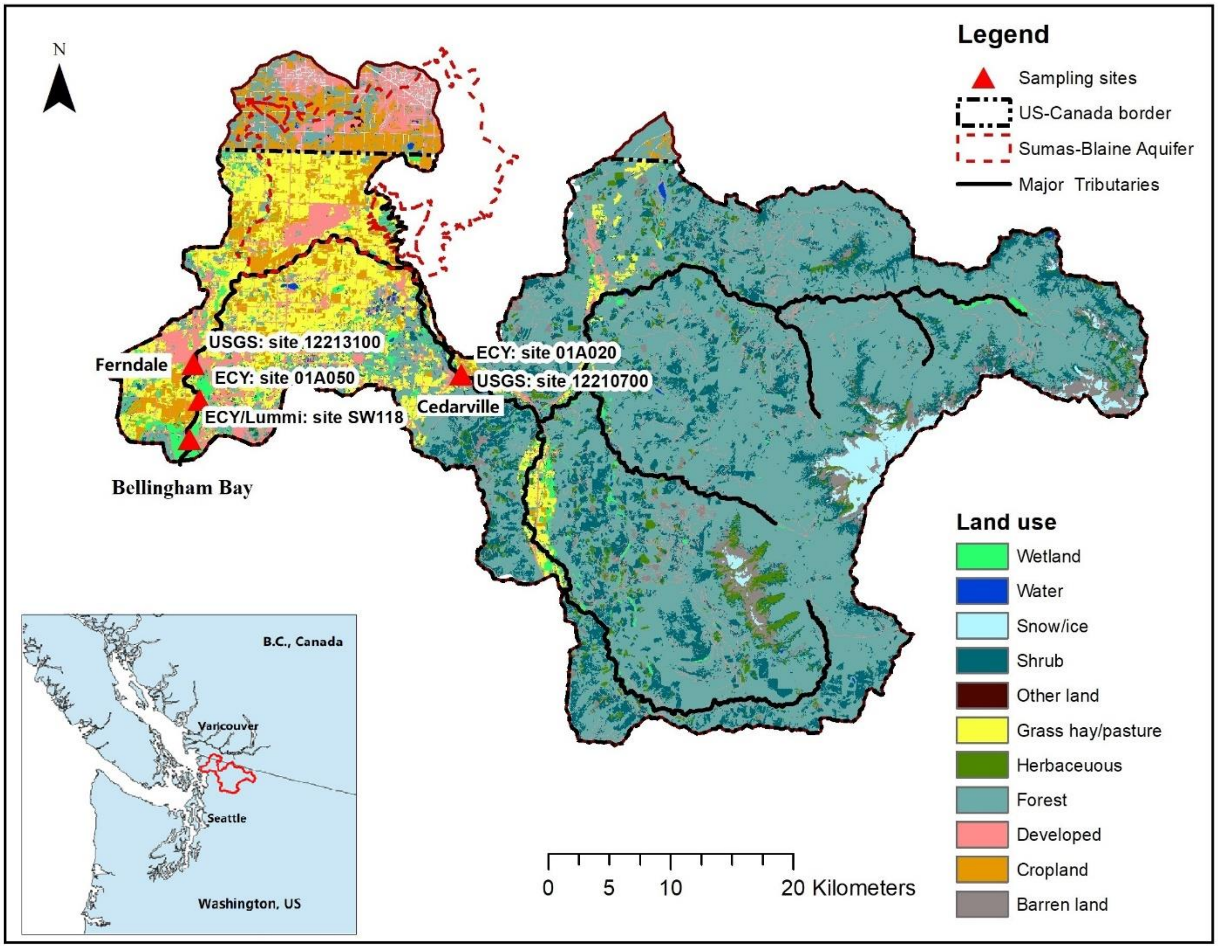

Figure 1: The Nooksack River Watershed (NRW): Land use and its major tributaries. The Sumas-Blaine Aquifer (SBA) underlays part of the agricultural land of the NRW in both the US and Canada. Gaging station measurements and years of data are as follows: USGS Site 12213100: Daily discharge (1977-2018) and TKN concentration (1995-1998); ECY Site 01A050: Nitrate concentration (1977-2016); ECY/Lummi Site SW118: TKN concentration (2001-2018); USGS Site 12210700: Daily discharge (2004-2018); ECY Site 01A120: Nitrate concentration (1977-2016).

3 Methods

For this study, the US portion of the watershed will be referred to as US-NRW, and Canadian portion as Canada-NRW. Due to disparity in accessibility and forms of data between the two countries, and because of differences in their agricultural practices (e.g., animal and crop types, and regulations), several major N fluxes in the US and Canadian portions of the watershed were calculated separately using different approaches. Most of the budget results for Canada-NRW were extracted from an existing nutrient budget model and N assessment for the Lower Fraser Valley in BC that included the Canada-NRW (Bittman et al., 2019), except for the following: atmospheric deposition, food import, human waste, and food waste. For these fluxes, the same approaches and assumptions were applied to calculate these components for both the US and Canadian portions of the watershed.

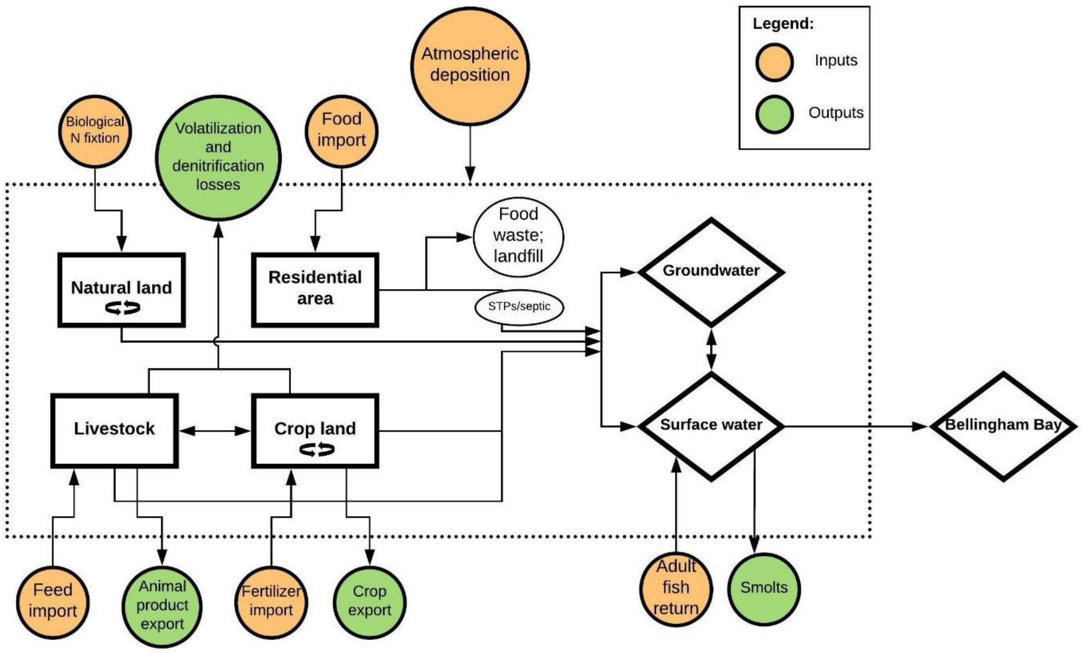

Figure 2: Budget components and N fluxes estimated in the Nooksack River Watershed. Dotted line: Watershed boundary; Circles: N inputs (orange) and exports (green) of the watershed; size of the circles is not indicative of the flux magnitude. Squares: Watershed land components; Diamonds: Hydrological components; Internal cycles in the natural and agricultural soils involving mineralization, nitrification, immobilization, and uptake.

Table 1: Budget Components and Data Sources (for all US Components and Some of the Canadian Components) for Nooksack River Watershed.

We integrated data from federal and state agencies and local agriculture experts with modeling results and literature values to quantify N fluxes in the NRW (Table 1). Fluxes were divided into three categories associated with input, export, and internal processes, respectively (Figure 2). More details of these methods can be found in the Supporting Information (SI). We also calculated watershed N retention and use efficiencies. We used 2014 as our target year because it was the year with the most available monitoring and survey data. When data were not available for 2014, we used data from the closest year available (Table 1).

3.1 N inputs

External N inputs to the watershed include: atmospheric deposition, food import for human and pets, feed import for farm animals, fertilizer import, and biological nitrogen fixation. Adult anadromous fish returning from the Pacific Ocean to the NRW were also calculated as an N input from outside the watershed (Figure 2).

Atmospheric deposition

Atmospheric deposition of total N and different forms of N in the whole watershed was extracted from simulation results of the Community Multiscale Air Quality Modeling System (CMAQ v5.2.1; https://zenodo.org/record/1212601) (Appel et al., 2017) at 4x4 km grid resolution. Meteorology was generated using the Weather Research and Forecasting model (WRF) (Skamarock et al, 2008). The Environmental Policy Integrated Climate (EPIC) model was used to provide land use and management data to CMAQ. The CMAQ and EPIC model simulations were conducted for our study region at the National Exposure Research Laboratory at EPA using specialized emissions inputs generated by Washington State University and emissions for CAFOs from Environment Canada (Bittman et al., 2019). More details of the air quality modeling can be found in Table S1.

Food import for humans and pets

Food consumption by humans was calculated based on census data in both countries (Supporting Information, SI) and per capita consumption of N in food (4.7 kg N yr^{-1}). The average per capita estimate of the county was made using nutritional data by human age classes for protein (USDA & HHS, 2016). We assumed that all food was imported into the watershed as suggested by local agricultural experts. Canadian population and household census data for BC subdivisions was downloaded and clipped to Canada-NRW boundary in ArcMap 10.7 (ESRI, 2011). Human food import was then calculated assuming 60% of available food N was consumed and 40% was not as a result of spoilage and wastage (Hall et al., 2009). Food import for pets was also calculated based on population and nutritional needs of dogs and cats. US Pet Ownership Statistics (AVMA, 2012) showed 37% of US households (census data) own dogs and 30% cats. These pet ownership values were assigned to Canada-NRW as well. Pet N consumption was calculated by converting average body weights to energy needs then further to nutrition intakes (Table 1). We assumed the average body weights to be 20 kg for dogs and 3.6 kg for cats (Baker et al., 2001).

Feed import for farm animals

For US-NRW, feed import was calculated as the difference between total N required by farm animals and local feed production, with the former calculated as the product of animal numbers and their nutritional needs. Dairy animal populations were estimated based on data from the Washington State Department of Agriculture (WSDA) dairy inspection program for 2014 (WSDA, 2018). We downscaled USDA census data at the Whatcom county level to estimate population data for other animals. Data were downscaled based on the proportional agricultural land of the county falling within the NRW boundary. Information on the daily N intake by lactating cows was provided by local experts. Nutritional needs for other animals in US-NRW were retrieved from various primary sources (Table 1). Feed import to Canada-NRW was downscaled from the existing Lower Fraser Valley grid model quantifying nutrient flows (Bittman et al., 2019), where data on animal population and local feed acreages were derived from Census of Agriculture and the BC Ministry of Agriculture (Bittman et al., 2019).

Fertilizer import

For US-NRW, we calculated imported synthetic fertilizer as the difference between N ‘requirement’ of each crop and available local manure. The N ‘requirement’ term described total crop uptake of fertilizer N (both synthetic fertilizer and manure) after various losses, and was calculated as:

$$\boldsymbol{N_{crop,rqr} = \sum_{i=1}^{i} A_i \times F_i \,/ f \qquad Eq.\,1}$$

where $\boldsymbol{N_{crop,rqr}}$ is the total crop N need (kg N yr^{-1}) in the watershed; $\boldsymbol{A_i}$ and $\boldsymbol{F_i}$ are respectively the planting area (ha) (WSDA, 2015) and recommended uptake N (kg N ha^{-1} yr^{-1}) of crop i, based on suggestion from local expert and extension documents (Table S2); $\boldsymbol{f}$ is a fertilizer coefficient that converts crop uptake N to total required fertilizer N (both synthetic fertilizer and manure) by factoring various losses under local conditions (Table 1; Supporting Information).

In Canada-NRW, fertilizer N import was extracted from the Lower Fraser Valley model, where fertilizer application was summarized from weekly application data collected from industry experts and farm surveys (Bittman et al., 2019).

Biological N-fixation (BNF)

For US-NRW, alder N fixation could be a substantial natural N source in the Northwest region (Compton et al. 2003; Wise & Johnson, 2011) and was calculated using the approach developed by Lin et al. (2019). A conservative annual fixation rate (100 kg ha^{-1} yr^{-1}) (Binkley, 1994) was multiplied by total alder basal area (ha), extracted from the Gradient Nearest Neighbor Structure map (Ohmann et al., 2011). Alder N fixation was not calculated for the Canada-NRW because tree species data were not available. Agricultural N fixation was not calculated because the area lacked major N fixing crops such as alfalfa, soybeans or leguminous cover crops.

Anadromous fish return

Return of adult anadromous salmonids from the ocean to their natal rivers and streams to spawn and die has historically been a source of marine-derived nitrogen to freshwater and riparian habitats in the Pacific Northwest (Compton et al., 2006; Gresh et al., 2000; Janetski et al., 2009). Current salmonid populations in Salish Sea watersheds are far below historical levels (Gresh et al., 2000). While some stocks are healthy, others are listed as threatened by the US Fish and Wildlife Service, and others are supported mainly by hatchery operations (Puget Sound Partnership 2017, https://www.psp.wa.gov/salmon-recovery-watersheds.php). We calculated the 2014 N input to the NRW from returning salmon and steelhead as a function of fish population, body mass, and the N content of the fish. Average body weights and N contents of fish were the mean values from regional literature values (Table 1). Fish populations were derived from spawning ground escapement estimates provided by the Nooksack Stock Assessment.

3.2 N outputs (exports)

N outputs included riverine export, ammonia (NH_3) volatilization, denitrification loss, and animal and crop product export. In this study, we also included N export from smolt migration out of the watershed (Figure 2).

Riverine export

The US Geological Survey (USGS) Load Estimator model (LOADEST) (Runkel et al., 2004; USGS, 2013) was used to simulate riverine transport of nitrate N at two locations (Figure 1): The upstream location (Cedarville) represented the upland watershed, which was predominantly forest (> 95%); model input data were daily discharge measured by USGS (Site 12210700, 2004-2018; Figure 1) and monthly nitrate concentration measured by Washington State Department of Ecology (ECY; Site 01A120, 1977-2016; Figure 1). The downstream location (Ferndale) near the mouth of the River represented export from the whole watershed; daily discharge was measured by USGS at Ferndale (Site 12213100, 1977-2018; Figure 1) and nitrate by ECY at nearby Brennan (ECY; Site 01A050, 1977-2016; Figure 1). Nitrate flux contributed by the lowland watershed was calculated as the difference between the whole watershed nitrate flux and upland nitrate flux.

Total Kjeldahl N (TKN, total organic N + total ammonia N) flux was estimated for the Ferndale location by LOADEST simulation using daily discharge data measured by USGS at Ferndale and concentration data measured by both USGS (Site 12213100, 1995-1998; Figure 1) and a collaboration between ECY and the Lummi Nation (Site SW118, 2001-2018; Figure 1).

Volatilization and denitrification losses

We calculated manure NH_3 volatilization in US-NRW based on National Resources Conservation Service (NRCS) estimates for Western Washington and information from local agricultural experts (Table 1 & SI): We assumed 35% pre-application volatilization loss during manure storage and housing; of what was applied in field, we assumed an average of 15% volatilization loss for both manure and synthetic fertilizer (Carey & Harrison, 2014; USDA-NRCS, 1998). Volatilization in Canada-NRW was extracted from the Lower Fraser Valley model results based on proportional agricultural land area. Denitrification loss was estimated to be 10% of applied manure and synthetic fertilizer in the entire NRW (USDA-NRCS, 1998). Denitrification in natural lands was not calculated and assumed to be part of N retention.

Crop product exports

In US-NRW, crop removal of N was calculated based on crop removal rate (extension documents, local expert, survey, and scientific literature, see Table 1), crop N content, and crop area (WSDA), as shown in Eq. 2:

$$\boldsymbol{N_{crop,rmv} = \sum_{i=1}^{i} A_i \times Y_i \times (1 - m_i) \times n_i} \qquad \boldsymbol{Eq. 2}$$

where $\boldsymbol{N_{crop,rmv}}$ is the total crop removal of N (kg N yr^{-1}) of the watershed; $\boldsymbol{A_i}$ and ; $\boldsymbol{Y_i}$are respectively the planting area (ha) and yield (kg crop mass ha^{-1} yr^{-1}) of crop i; $\mathbf{m_i}$ is the moisture content (%) of crop i, and $\mathbf{n_i}$ is the N content (%) of crop i on a dry weight basis. N export in crop product for Canada-NRW was derived from the Lower Fraser Valley model, where crop export was computed as harvest removal in berries (raspberries and blueberries), the dominant export cash crop in this part of Canada (Bittman et al., 2019). Export of forest product was not calculated.

Animal product exports

Because most of the milk, eggs and other products produced in the US-NRW were not consumed locally, animal product export was calculated as the sum of N in milk and eggs and N export of live animals. Data on production rates, animal populations, and N contents were from USDA and WSDA (Table 1). Animal product export of N from Canada-NRW (mainly N in meat and milk products from poultry, dairy, and pigs) was estimated from downscaled data from the Lower Fraser Valley model results on milk and meat N exports.

Smolt export

Smolts are juvenile salmon migrating from rivers to the ocean. We included smolts from both natural and hatchery origins. Smolt mass was estimated based on fork length (the length of a fish from its nose to the middle caudal fin rays) data and length-to-weight equations. Smolt mass was then coupled with N content and population data to calculate total smolt N export. Data and equations were provided by the Skagit River System Cooperative, the Lummi Nation, and literature review (Table 1).

3.3 N internal processes

Sewage treatment plants and septic export

We treated most of the N fluxes in sewage treatment plants and septic systems as internal transfers under the assumption that releases from these sources either went into soil and/or groundwater retention, or surface water fluxes (Figure 2). Total nitrogen (TN) load from sewage treatment plants draining to the Nooksack River was calculated as the product of observed and extrapolated effluent discharge and TN concentration (SI). When a measurement was missing at certain sewage treatment plants, TN load was extrapolated based on the population size served. There was no sewage treatment plant outlet within the Canada-NRW boundary, therefore, sewage effluent in Canada-NRW was counted as N export that left the watershed. To estimate septic inputs to the whole watershed, the population not on sewage was multiplied by an average per capita waste rate (4 kg N yr^{-1}) (USEPA, 2002) and 91% septic leaching rate (USEPA, 2002). In US-NRW, the ratio of population on sewage and population on septic system was about 2:3. The same ratio, which was also applied to Canada-NRW.

Food waste

Food waste was estimated to be 40% of the available food supply based on Hall et al. (2009). We assumed that all food waste was part of N retention and went to landfills, which in the long term can be subject to volatilization and/or other losses that we were unable to quantify in this project.

Crop application of dairy manure Annual manure application was calculated based on animal populations, excretion rates, and pre-application emission losses. In US-NRW, the total crop N 'requirement' and proportional application of manure vs. synthetic fertilizer were provided by local farmers for each crop type. Pre-application volatilization loss was taken into account to calculate total manure required. This value was then compared with dairy manure excreted to decide if there was a net import or export of manure fertilizer. Manure application in Canada-NRW was extracted from the Lower Fraser Valley model, where excretion rates were computed as the difference between N fed based on industry data and N in animal products (Bittman et al., 2019).

Crop to animal feed

We assumed all the feed crops were retained in the watershed and used as local animal feed. Local production of silage corn and grass hay provided about 50% of the dry matter required by lactating cows, with the other 50% of their feed was imported as soybean and alfalfa required for milk production. The remaining US-NRW feed crops were used to feed other livestock. In Canada-NRW, all local feed was consumed by dairy cows based on the Lower Fraser Valley model (Bittman et al., 2019), and thus we calculated local feed as the difference between total feed required and the imported feed for cows (Bittman et al., 2019). Total feed required was estimated based on surveyed cow populations and their nutritional needs for N, and the proportion of feed from import was acquired from a previous survey (Bittman et al., 2019; Sheppard et al., 2010).

3.4 N retention and use efficiency

N retention was defined as the amount of annual N inputs remaining in the watershed after accounting for removal via known pathways such as riverine, gaseous, and agricultural exports. Fates of N retention include storage in plant and animal tissues, soil and groundwater, and landfill, but may also include unaccounted losses.

We calculated crop N use efficiency (NUE) as the ratio of crop N harvest removal and the sum of manure and synthetic fertilizer N applied. We also calculated NUE for production in the whole watershed using two methods: 1) the farm-gate method calculated NUE as the ratio of N removed off-farm in products vs. total N inputs to the entire watershed (Ovens et al, 2008), and 2) 'commercial' whole-farm NUE was the ratio of N in crop and animal products over the import of feed and fertilizer N only (Bittman et al., 2016). The crop NUE helps interpret efficiency of cropping systems and potential losses, though losses to other pools (e.g., ground and surface water) are not explicitly separated from N storage in soils and plant parts not removed in harvest (residues, root tissue, etc.). Farm-gate NUE provides critical information on both agronomic efficiency and environmental risks for the whole watershed, and has been used as a policy instrument and the basis of regulation of farm nutrient levels and losses (Ovens et al, 2008; van der Meer, 2001). The 'commercial' whole-farm NUE method excludes 'free' N inputs and mitigates the need to account for inputs beyond the farmers control such as deposition and fixation (Bittman et al., 2016; Buckley et al., 2016). It also helps with the assessment of economic consequences.

4 Results

4.1 N inputs

N imported as animal feed and synthetic fertilizer was about 8,600 tonnes N yr^{-1} in total and contributed 81% of N influx to the watershed (Figure 4; Table S3). The largest influx of N was animal feed accounting for about 58% of N inputs to the watershed. Feed imports supporting dairy and poultry production were nearly equal for the entire watershed, accounting for 30% and 29% of total NRW input, respectively, with most of the dairy production on the US side, and much of the poultry production on the Canadian side (Table S3). In the US-NRW, imported feed for dairy cows was more than 3,100 tonnes (metric ton) N yr^{-1}, making up 42% of US-NRW N input. In the Canada-NRW, annual dairy feed import was about 21 tonnes N yr^{-1} representing < 0.7% of Canada-NRW N input, while imported feed for poultry was over 2,400 tonnes yr^{-1} representing 78% of Canada-NRW N input. On the watershed level, annual import of over 2,200 tonnes synthetic fertilizer was the second largest N source representing 21% of total input. About 57% of imported fertilizer was applied to feed crops (grass hay and corn silage) and the rest was applied to other crops.

Other sources of N contributed approximately 19% of N inputs to the watershed (Figure 3&4). Atmospheric deposition contributed 10% to the total N input: about 4% was deposited on urban and agricultural lands and 6% was deposited on upland forest. Food imports for humans and pets contributed about 8% and 1%, respectively. Alder fixation and marine-derived return of adult anadromous fish both represented about <0.07% of N inputs each. Smaller amounts of these inputs in Canada-NRW than US-NRW arose because of smaller proportions of land area and total population in the former than the latter (Figure 3).

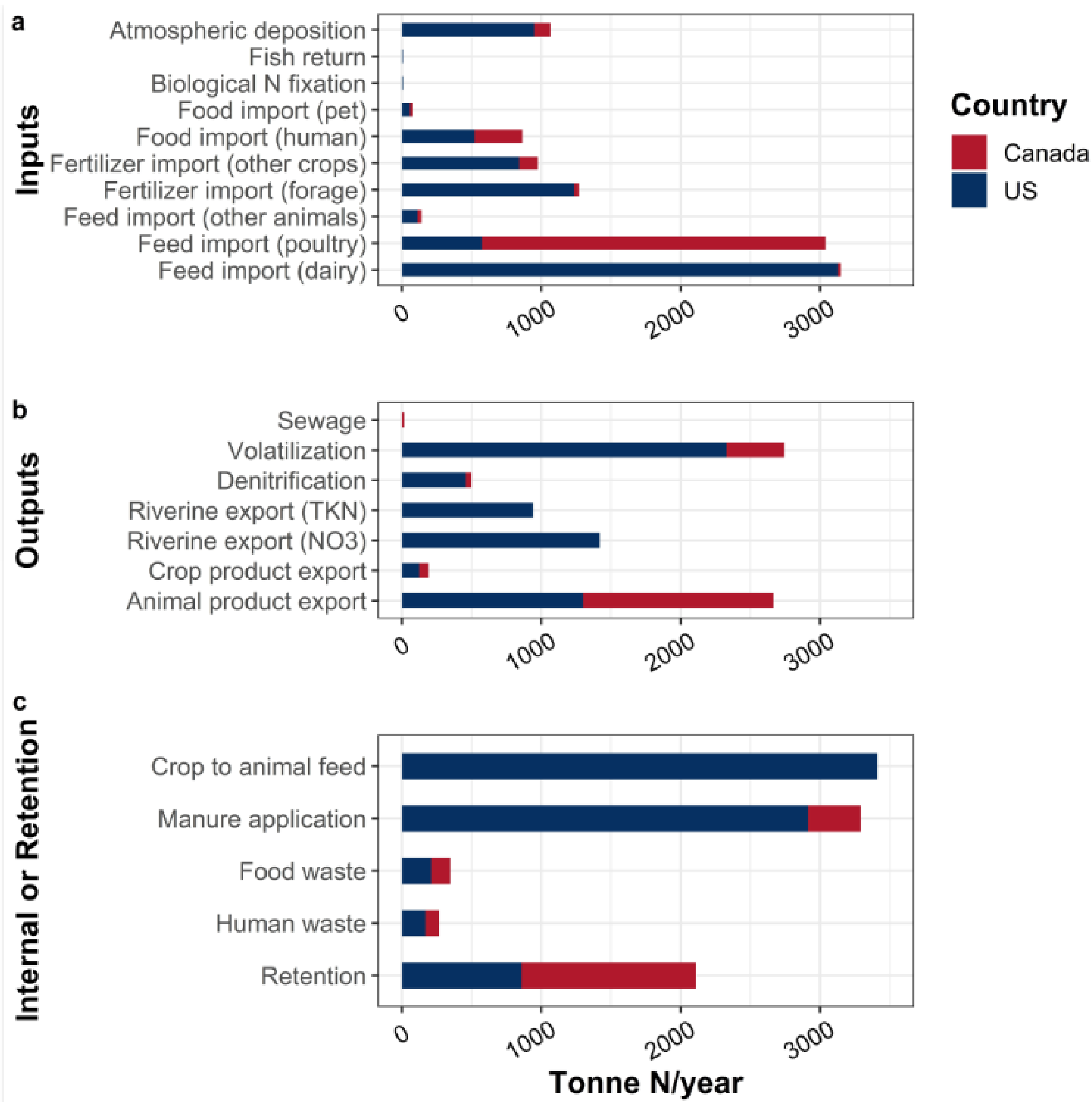

Figure 3: Annual N fluxes of the Nooksack River Watershed (NRW). a. Inputs; b. Outputs. 'Sewage' refers to N in the effluents from sewage treatment plants in Canada-NRW that drained out of the watershed; c. Internal fluxes or N retention. Retention includes storage in groundwater, soil, biomass and unaccounted N losses. 'Human waste' refers to N in sewage effluents in US-NRW and septic fluxes in both Canada-NRW and US-NRW. Forage is defined as crops for animal feed.

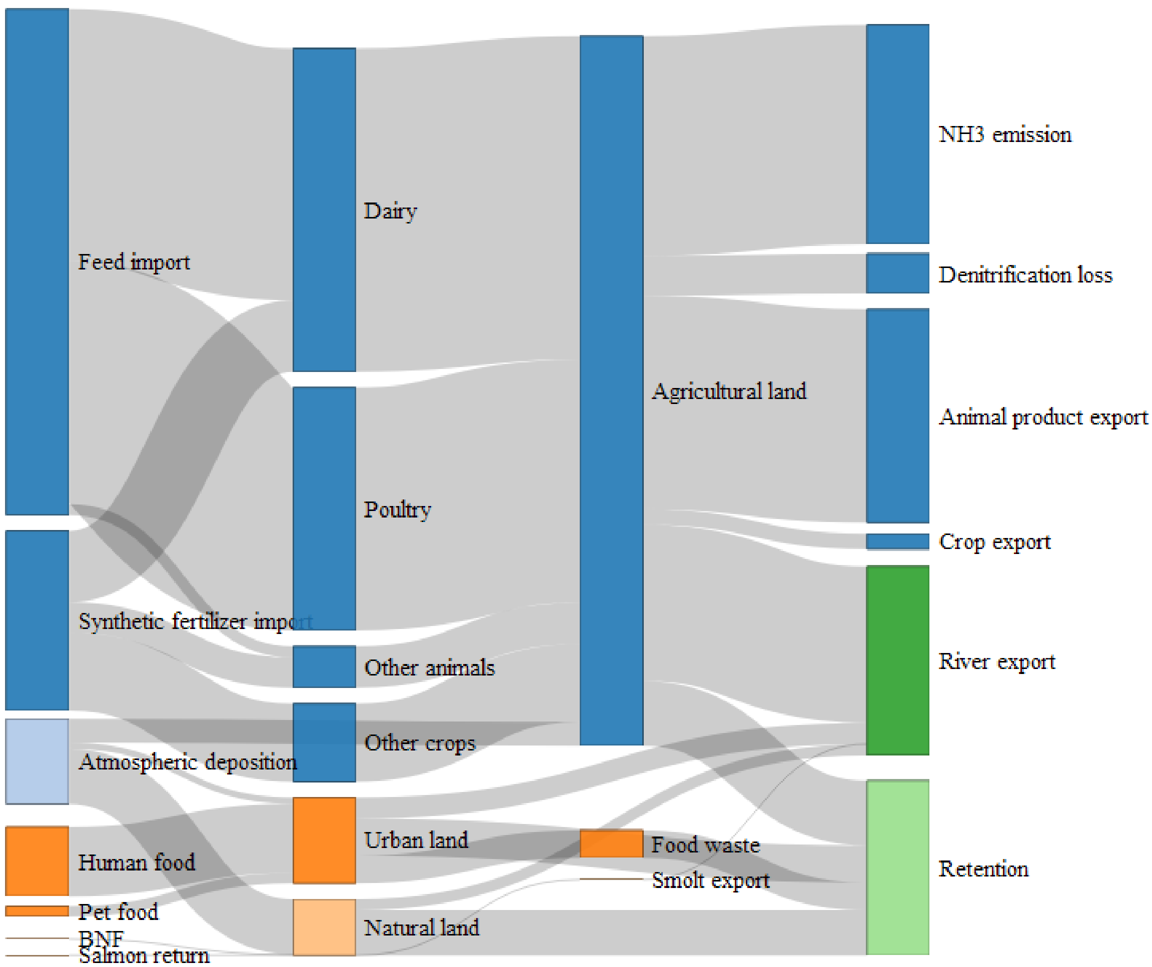

Figure 4: N fluxes in the Nooksack River Watershed. Grey bars represent N fluxes flowing from external inputs on the left side to internal cycling in the middel and then export/storage on the right side. Different colors represent different sectors – dark blue is agriculture, light blue is deposition, orange is residential fluxes and green represents river export and retention within the basin on an annual timestep. Bar height is proportional to the magnitude of the N flux.

4.2 N exports

The largest N export was NH_3 volatilization loss, with an estimated 2,745 tonnes N yr^{-1} and 32% of total export (Figure 4; Table S3). Nearly three quarters of the NH_3 volatilization was associated with dairy manure in US-NRW, and most volatilization (78%) occurred during manure storage and housing processes whereas 22% occurred after field application. Volatilization loss associated with poultry manure application was only 10% of total volatilization loss. Export of N in animal products was the second largest flux from the watershed, contributing 2,666 tonnes N yr^{-1} or 31%. Milk was the primary product in US-NRW and poultry products (meat and eggs) dominated Canada-NRW export (Table S3). Denitrification (as N_2 and N_2O) associated with the application of manure and synthetic fertilizers accounted for 6% of total N export. In comparison, crop export was relatively small accounting for only about 2% of N export, with 126 tonnes N from exporting various horticulture crops in US-NRW and 64 tonnes N from berry production in Canada-NRW annually.

Hydrological export was another major pathway for N leaving the watershed, responsible for 28% of all N loss. In 2014, the Nooksack River transported 1,420 tonnes of NO_3-N and 940 tonnes of TKN into Bellingham Bay (Figure 3). Nitrate was thus approximately 60% of the riverine N export. The upland watershed (Cedarville) contributed about 750 tonnes NO_3-N, 53% of the total riverine nitrate export. Export of N via smolt migration out of the watershed was less than 0.001% of the total N export.

Table 2: Nitrogen use efficiencies (NUEs) in the Nooksack River Watershed

4.3 N retention, internal fluxes, and nutrient use efficiency (NUE)

The watershed N balance or N retention, calculated as the difference between inputs and exports, was about 2,130 tonnes N or about 20% of total N inputs (Figure 3; Table S3). This may include potential losses to groundwater, which we did not quantify. The 20% N retention may include accumulation in biomass of perennial crops (e.g., berries) and natural vegetation (e.g., forests), in biomass of animals (humans, pets, and stock), and in soils. It also may include other fluxes such as NO_x emission and natural denitrification that we could not quantify at this scale.

The largest internal N fluxes were associated with dairy production. Locally-grown feed provided about 1,767 tonnes N to dairy farms in US-NRW to support forage production for nearly 35,000 cows. About 2,554 tonnes N from dairy cow manure was applied on the US side. N fluxes associated with dairy cows on the Canadian side were smaller than their US counterparts and much lower than poultry manure application. In Canada-NRW, local crop feed provided 100 tonnes N for 1,323 dairy cows, and about 96 tonnes N from dairy manure was applied to crops; in comparison, 269 tonnes N in poultry manure was applied to crops. Internal N flux associated with human waste was only 2.4% of inputs or 265 tonnes in the NRW. Unexpectedly, estimated N flux in food waste was 346 tonnes for the entire watershed, slightly higher than N flux in human waste.

Crop NUE for the entire watershed was 51% regarding total manure and fertilizer (pre-volatilization loss). However, crop NUE for the watershed was at 67% for applied manure and fertilizer, and was greater in the US-NRW (71%) than in Canada-NRW (31%). Using the farm-gate method (Ovens et al., 2008), we estimated that about 27% of total N input to the entire watershed was transferred into final crop and animal products, all of which was exported from the watershed (2,860/10,594 tonnes). Because all animals were transported elsewhere for slaughter, we assumed no N retention due to slaughtering and rendering processes. Using the 'commercial' whole-farm method (Bittman et al., 2016), we found the crop and animal product export equaled 33% of feed and total fertilizer inputs for the entire watershed, 24% for US-NRW (1427/5889 tonnes), and 53% for Canada-NRW (1433/2685 tonnes). In addition, animal products equaled about 36% of total feed and fertilizer (for feed crops) inputs for the watershed, with poultry products accounting for 43% of poultry feed import and milk export accounting for 29% of feed and fertilizer inputs to the dairy system.

5 Discussion

5.1 Inputs and internal cycling

N inputs were high on both the Canadian and US portions of the watershed. Input rates averaged about 50 kg N ha^{-1} yr^{-1} across the entire NRW, comparable to the state of California and the entire US (45 kg N ha^{-1} yr^{-1}), but smaller than the Netherlands or China (100 kg N ha^{-1} yr^{-1}) (Liptzin and Dahlgren 2016). Most of these inputs were concentrated in the lower valley of the NRW (23% of the area). N imports were largely related to agriculture, primarily animal production – either directly as feed for poultry (Canada) or dairy cows (US) or as fertilizer for cow forage. Much of the manure produced by animals was applied to crops in both the US and Canadian portions of the watershed. This application reduced the need for synthetic fertilizer and provided an important way to recycle feed N within the watershed. However, given the quantities of N imported for animal feed, application of substantial quantities of manure to the relatively small land base provided opportunity for inadvertent N losses. Relative to other sources of externally derived N, background sources (i.e., N fixation in natural lands and salmon returns) were each <1% of inputs at the NRW scale.

The high inputs were used with relatively low efficiency (Table 2). The 67% crop NUE for the watershed regarding applied manure and fertilizer was lower than US national average crop NUE of 70% (Zhang et al., 2015). Crop NUE was higher in US-NRW than in Canada-NRW (Table 2). This could be caused by the relative amounts of manure applied. US farmers have higher numbers of regulations and rules under local and state efforts to reduce the agricultural loading of nitrate to the environment (Cox et al., 2005), therefore they were more likely to follow extension recommendations on fertilization rates. Another major reason for different overall crop NUEs was based on crop types. Berries were the dominant crop types in Canada-NRW and had low N content of about 0.1% in the exported fruits, while forage grass and corn, harvested 4-6 times per year as dairy feed, had N content as high as 3%. Crop NUE alone does not provide comprehensive information on farming activities, and should be combined with other criteria such as farm-gate NUE to understand management efficiency.

In contrast to crop NUE, both the farm-gate and commercial whole-farm NUEs were higher in Canada-NRW than in US-NRW (Table 2). This could be attributed to a higher feed to animal product ratio of the US-NRW dairy system (2.7:1) compared to that of the poultry production in Canada-NRW (2:1). A recent Lower Fraser Valley study showed that using rendering products as poultry feed was a very effective reuse of local N and could improve NUE in British Columbia (Bittman et al., 2019), but using rendering products is prohibited in dairy production due to health concerns. In addition to animal types, stocking rate can also have important consequences for NUEs (Powell and Rotz, 2015): the dairy stocking rate in Canada-NRW was about 1 cow acre^{-1}, whereas in US-NRW it averaged 1.3 cow acre^{-1} for all forage crop land and 1.8 cow acre^{-1} for some crop land with high management intensity where most dairy cattle were kept. The farms with lower stocking rate required less feed import since local feed production was sufficient, which resulted in higher whole farm NUEs (Bittman et al., 2019).

5.2 Release of N to the environment

Loss of over 50% of N inputs to the environment, primarily as volatilized ammonia and hydrological N exports to surface water and groundwater, has a strong potential to adversely affect human health and the environment (Townsend et al., 2003). Ammonia, predominantly from losses related to housing and storage of manure, can contribute to regional smog and odor problems (Barthelmie & Pryor, 1998; Kotchenruther & Taylor, 2014), and can harm human respiratory health (Paulot & Jacob 2014) Enhanced N deposition resulting from elevated N emissions can cause significant damage to terrestrial and aquatic ecosystems, including cation leaching, altered nutrient stoichiometry in streams and lakes, and changing biodiversity (Clark et al., 2018; McMurray et al., 2013).

Annual riverine N export for the NRW was 28% of total N input, which may contribute to current and future eutrophication and hypoxia in Bellingham Bay (Khangaonkar et al., 2019; Mohamedali et al., 2011). TKN accounted for 40% of NRW riverine N export, indicating substantial surface input from organic N and ammonia, potentially originating from soils rich in organic matter and anthropogenic N (Bronk et al., 2007; Kroeger et al., 2006). Hydrologic export that primarily occurs during the cool, wet seasons when there is low biotic removal potential poses a substantial challenge to nutrient management (Compton et al., 2019; De Girolamo et al., 2017; Welter & Fisher, 2016). Wet season precipitation and rising groundwater levels were also linked to high seasonal soil nitrate concentration, which could lead to elevated N loading to ground waters and high nitrate levels in the aquifer (Carey, 2017; Cox et al., 2018).

Both the forested upland and the agriculturally influenced lowland make substantial contributions to the riverine N export. The lowland comprised 24% of the entire watershed, was 66% agricultural land, and contributed 47% of the riverine NO_3-N export. The upland watershed comprises 76% of the whole watershed, was >95% forest, and contributed 53% of the riverine NO_3-N export. Forest edges, which have been increasing as a result of forest fragmentation, may function as nutrient traps and concentrators (Weathers et al., 2001), particularly for ammonia emissions. This phenomenon may influence the forest riverine N export. Our results indicated the importance of forest management to downstream water quality and nutrient balance.

We did not directly quantify N flux to groundwater due to its complexity and instead included it as part of watershed N retention, but we acknowledge that some portion of the N applied leaches into groundwater. For example, rates of nitrate leaching from the soil were substantial below raspberry fields in the area (80-240 kg N ha^{-1} yr^{-1}) (Loo et al., 2019). Combining crop area data with published soil nitrate data in this area or in watersheds with similar land use and weather, we did a back-of-envelope estimation of the range of N flux leaching under different land uses. Based on nitrate leaching rates under raspberry field (Loo et al., 2019) and post-harvest soil survey of different crops in South Abbotsford and West Sumas (Sullivan & Poon, 2016), townships that are located in northern NRW, we estimated about 260 - 430 tonnes N entered the groundwater annually in Canada-NRW, assuming about 80% of post-harvest soil nitrate-N was lost to leaching (Carey, 2002). Previous studies showed that nitrate leaching following dairy manure application on forage crop land ranged between 32 and 153 kg N ha^{-1} depending on fertilization rate (Demurtas et al., 2016; Paul & Zebarth, 1997; Tarkalson et al., 2006). Hence, potentially there was about 930-1,100 tonne N leaching under forage crop land in US-NRW, given that 70% of the forage land there was managed with high intensity. This represents about 9-10% of all N inputs. These results cannot be viewed as a complete quantification of groundwater N flux in the watershed, yet they provide insights about the potential N contamination of groundwater. We estimated that N loss to groundwater could represent about 56-72% of N retention in the NRW.

Much of the applied N could be incorporated in soil organic matter and remain in the soil for many years to contribute to future risk of contamination of water resources (Sebilo et al., 2013). Studies have shown that legacy nutrients can become a dominant and long-term (>10 yr) source of excess nutrients in many intensively managed watersheds (Chen et al., 2018; van Meter et al., 2016). Groundwater N might eventually contribute to surface water export over time, directly through irrigation using groundwater or indirectly as the groundwater flowpaths emerge in streams.

Even though the study area contained a small portion of urban land, management of food and food waste could represent an opportunity to reduce N loss based on our budget results. The food waste portion was slightly greater than the sewage treatment plant contribution in the watershed (Figure 3). Some of the negative impacts of excess N due to food production could be partially addressed by reducing food waste and dietary N footprints in urban areas (Shibata et al., 2017), which must include community collaboration. For example, systems thinking can support an integrated agricultural and food system to optimize food utilization, and technologies can help improve the efficiency of using food waste for biogas and compost (Halloran et al., 2014). These efforts need to be promoted through partnerships among the government, society groups, and industry

5.3 Implications for effective N management

Enhancing both dairy and cropping efficiencies are vital to achieving effective nutrient management (Harrison, 2007). On the US side of the watershed, there have been many conservation efforts by local and state agencies aiming at improving N management efficiencies and reducing agricultural loading of nutrients to the environment. For example, Whatcom County adopted a Manure Control Ordinance that restricted field manure application timing for forage production to April through September to reduce leaching during wet seasons (Cox et al., 2005). Whatcom Conservation District and USEPA developed a Progressive Manure Application Risk Management (ARM) System, a decision-making tool using real-time field and weather information, to help guide manure applications and reduce manure losses (Embertson, 2016). Washington State also mandated the development of Nutrient Management Plans for all dairy farming operations that handled more than 700 dairy cattle. These initiatives may lead to important reductions in N release to the environment for the NRW in the future.

As the major N loss pathway in the NRW, ammonia emission is controlled by multiple factors such as livestock and manure management systems (Sanchis et al., 2019). Previous research found that it was necessary to shift from single-stage emission abatement options towards a whole-chain perspective (Sajeev et al., 2019). In the NRW, livestock housing and storage was a major source of ammonia emission. Reducing volatilization loss during this stage can be achieved by quantitatively understanding of the effect of temperature, wind speed, relative humidity and ventilation rate on ammonia release rates from dairy cattle housing (Sanchis et al., 2019). Moreover, multiple mitigation strategies can be combined at different stages (housing, storage, and application) to reduce overall whole-farm emission, for example, frequent removal of manure, anerobic digestion, and manure acidification were all found effective in reducing emissions (Sajeev et al., 2019). Adjusting cattle diet such as lowering dietary crude protein were also associated with decreases in ammonia emissions rates and emission as a percentage of N intake (Liu et al., 2017). Subsurface application of dairy slurry can also decrease ammonia volatilization compared to surface application (Saunders et al., 2012).

The potential contribution of nitrate leaching under agricultural land in the watershed is substantial. Increasing manure application rates were associated with higher leaching in the dairy system in the region (Hill, 2013; Paul & Zebarth, 1997) To improve N management on agricultural lands in this area, efforts should not be limited to forage crops that were most commonly associated with dairy farms, because high leaching rates were also measured under berries and vegetable crop lands (Loo et al., 2019). Nitrate leaching under the same land use can also vary widely in response to variations in climate factors, management practices and soil properties (Loo et al., 2019). Different N treatments can be imposed on cropping systems to reduce nitrate leaching. For example, the use of nitrification inhibitor dicyandiamide and/or biochar was found successful in reducing nitrate leaching (Di & Cameron, 2002; Lehmann & Joseph, 2009). Switching fertilization types (such as using compost) can also help reduce leaching (Basso & Ritchie, 2005). There were also seasonal variations: nitrate leaching during the growing season may be minimal compared to leaching losses that occur between the harvest of one crop and the planting of the next (Basso & Ritchie, 2005). Cover and relay crops could help minimize N leaching during the winter depending on conditions (van Vliet et al., 2002). Any nutrient reduction strategies developed should account for the strongly seasonal hydrology of this area.

Integrated nutrient management should also focus on reducing imports and seeking export opportunities for excess nutrients. Harrison et al. (2012) suggested that the most effective approach should include accounting of managed nutrient imports and exports from the farm, and the estimation of on-farm excess (or deficits) of nutrients. Decreasing stocking rate (animal per unit of land) can help reduce imports of both fertilizer and animal feed. Higher animal stocking rates placed more challenges on nutrient management, since high animal densities resulted in higher expenses for feed import and also higher excretion rates and ammonia loss rates (Powell & Rotz, 2015). Lower stocking rates can also represent more land area being converted to agriculture, representing an extensification (van Grinsven et al., 2015). Planting N-fixing cover crops can also help reduce the usage and import of fertilizer. Transporting excess manure offsite to be used as fertilizer elsewhere can help with the overapplication issue and reduce emission and leaching losses.

Harrison et al. (2012) suggested that strategies and technologies to achieve N reduction vary in their degree of economic feasibility and environmental impact. Site-specific and cost-effective Best Management Practices (BMPs) can only be developed with the collaborations of farmers, agencies, and scientists. Continuous soil and groundwater monitoring programs can help establish quantifiable solutions. Temporary lack of water quality improvements cannot be interpreted as a failure of the BMPs without knowing the residence time of groundwater and associated soil conditions, because accumulated organic matter mineralizes gradually over time and can cause lags in soil and groundwater quality improvements (Carey, 2002; Sebilo et al., 2013; van Meter et al., 2016; Wassana et al., 2006).

5.4 N budget uncertainties

The integrative NRW-N budget helps us understand N cycling in the watershed and can be used as an environmental performance indicator to guide future nutrient management; Still, major uncertainties in our assessment could arise from several issues:

1) There was limited information about specific farm practices such as total manure application rates and methods on each farm, which was regarded as confidential business information. It may have resulted in inaccurate representations of the agroecosystems and nutrient flows into and out of the watershed (Oenema et al., 2003).

2) Even though we attempted to capture most key sectors in the NRW, we did not estimate N fluxes from some other components in the N cycle. For example, forest fertilizers on private land, seed inputs or N-containing deicer used at the Abbotsford airport in the Canada-NRW (personal communication: Environment and Climate Change Canada), or N influx from migrating birds. Where studied, these fluxes have generally been a small proportion of N input budgets (McBroom et al., 2008; Olson et al., 2005). We also may underestimate denitrification and volatilization losses by not accounting for emission sources other than fertilizer and manure.

3) Generalization about certain processes could result in further computational errors. For instance, we used average denitrification (10%) and volatilization (35% pre-application and 15% post-application) loss rates for manure and fertilizers for the entire US-NRW, even though they probably varied among fields in real practice due to variabilities in application method, timing, weather, soil, and other factors. Denitrification in manured soils in the Pacific Northwest can range between 5 to 30% (Paul & Zebarth, 1997; USDA-NRCS, 1998), and a 17% of annual denitrification loss was measured in BC dairy farms (Paul & Zebarth, 1997). Based on these assumptions, annual agricultural denitrification was estimated ranging between 220 and 1400 tonnes, with our current result being on the lower end. Similarly, volatilization loss in western Washington can range from 10 to 50% during storage and housing and from 5 to 30% after application (USDA-NRCS, 1998), representing a potential error ranging from -68 to 44% in our volatilization estimation.

4) Non-continuous water sampling and potential errors during sampling and flux simulation (LOADEST) could lead to deviation from the actual riverine N loads.

5) There were uncertainties associated with CMAQ and EPIC simulations. For example, meteorology in the region is challenging to model; CMAQ could underestimate deposition from fog in complex terrain such as the forested upland; fertilization rates for many local crops could be underestimated or overestimated in EPIC; also, EPIC did not account for manure that was generated and applied locally—ammonia emission from animal manure was simulated separately in CMAQ.

6) Lastly, as a bi-national study, resolving issues caused by differences in data collection and resolution between the two countries and the limit of our understanding of the transboundary ecosystem could contribute to uncertainties in our budget. Downscaling N

budget results from the Canadian Lower Fraser Valley model could have induced certain systematic bias and errors because of applying different boundaries.

Despite these limitations, we consider this budget to be a current best estimate of N inputs, export s and internal cycling using local data and knowledge—this type of budget is still rare for watersheds in the Pacific Northwest area (Swaney et al., 2018). The NRW N budget can provide a potential roadmap for prioritization of pathways to reduce N release to the environment.

6 Summary

Our nitrogen budget of the transboundary watershed helped to identify several key issues related to better N management. Nearly 81% of the N inputs to the basin were used to support agricultural production, most of which was animal feed import. Watershed N retention was about 20% of the total input. The largest export from the NRW was in the form of ammonia from the agriculture sector (32%), which could have air quality implications for local residents and surrounding areas. Riverine export of nitrogen in to Bellingham Bay was a substantial portion of the export (28%). While the climate and physiography are similar between the US and Canada in the NRW, the different sides of the border provide contrasts in N management and use efficiency: Crop NUE was higher on the US side of the watershed, but both the farm-gate and commercial whole-farm NUEs were higher in Canada-NRW. These differences were driven by the types of animals raised, manure management regulations and reporting, and farm economics. As might be expected, different policy frameworks had a large impact on key components of nutrient management in different portions of the watershed. We had several N fluxes that were difficult to quantify with the available information. Improved information will help close our knowledge gap in the future. Similarly, better quantification of N fluxes from the US to Canada (in airflow) and from Canada to the US (in surface and groundwater flow), will help provide better identifications of N imbalances, and thereby enhance strategic policy-making to address those challenges.

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available in Table S3 for review purposes. The views expressed in this article are those of the author(s) and do not necessarily represent the views or policies of the government agencies.

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Table 1. Budget Components and Data Sources (for all US Components and Some of the Canadian Components) for Nooksack River Watershed.

	Component	Parameter	Data source
INPUTS	**Atmospheric Deposition**	Total N deposition	EPA-CMAQ (Appel et al., 2017)
	Food Import (Human)	Human population	USDA-NASS, 2017: 2015 census
		Nutritional consumption, per capita	USDA & HHS, 2016; Hall et al., 2009
	Food Import (Pet)	Watershed household	USDA-NASS, 2017 (2015 census)
		Population and body weights: Dogs and cats	Dogs - 37% of watershed households; Cats - 30% of watershed households. Assuming one pet per household; US Pet Ownership Statistics (AVMA, 2012); Baker et al., 2001
		Nutritional and energy needs	Veterinary online manual (link); Pet Basic Calorie Calculator (link)
	Feed Import	Animal populations (other than dairy cow, such as duck, goat, turkey, hogs, sheep, etc.)	USDA-NASS, 2017: 2012 data
		Dairy cow population	WSDA (2018)
		Nutritional needs of farm animals	Boyer et al., 2002; Hong et al., 2011, 2013; National Research Council, 1994; Veterinary online manual (link); Nennich et al., 2005; Bittman et al., 2019; Goyette et al., 2016
	Fertilizer Import	Crop land	WSDA, 2015
		Crop fertilization rates	Local agriculture experts (personal communication: WCD); Lin et al., 2019; Oregon (link) and Washington (link) Extension online documentations
	Biological N Fixation	Alder density	Ohman et al., 2011
		Alder N fixation rate	Binkley, 1994
	Adult Fish Return	Salmon population and size	Nooksack Stock Assessment (personal communication: WDFW Fish Program)
		Adult fish body weight	Gresh et al., 2000
		Adult fish body N content	Moore et al., 2011

	Component	Parameter	Data source
OUTPUTS	**Riverine Nitrate/TKN Export**	Flow	USGS site 12213100 (USGS, 2016)
		Concentrations	Nitrate: WA Dept. of Ecology site 01A050; TKN: Lummi Nation site SW118; USGS site 12213100
		Natural land area	NLCD 2011 (Homer et al., 2015)
		Forest N leaching rate	Cole et al., 1992
	NH_3 Volatilization	Animal manure application rates	Bittman et al.., 2019; Hong et al., 2011, 2013; Nennich et al., 2005; Sheppard et al., 2011; USDA-NASS, 2017 (2012 data); WSDA (2018)
		Synthetic fertilizer application rates	Local agriculture experts (personal communication: WCD); Lin et al., 2019; Oregon (link) and Washington (link) Extension online documentations; WSDA, 2015
		Fertilizer and manure volatilization rate/percentage	Carey & Harrison, 2014; USDA-NRCS (1998)
	Denitrification Loss	Fertilizer and manure denitrification rate/percentage	USDA-NRCS (1998)
	Animal Product (Milk)	Dairy cow population	WSDA (2018) (2014 data)
		Milk N production rate	USDA-ARS, 2018 ; Bittman et al., 2019 ; Goyette et al., 2016
	Animal Product (Other)	Animal populations (other than dairy cow)	USDA-NASS, 2017 (2012 data)
		Animal product N content	USDA-ARS, 2018 ; Bittman et al., 2019 ; Goyette et al., 2016
	Crop Product	Crop land	WSDA (2015)
		Crop N content	USDA-NRCS, 2019
	Smolt Export	Smolt population and size	Lummi Nation (personal communication: Julie Klacan and Sandra O’Neil, Washington State Dept. of Fish and Wildlife)
		Smolt body weight equation	Skagit River System Cooperative (personal communication: Eric Beamer, SRSC Research Department)
		Smolt body N content	Moore et al., 2011

	Component	**Parameter**	**Data source**
INTERNAL CYCLING	**Human Waste**	Sewage Treatment Plants (STPs) monitored N in effluents	Everson STP (link); Lynden STP (link); Ferndale STP (link)
		Septic population: total population - service population on sewage	USDA-NASS, 2017; Everson STP; Lynden STP; Ferndale STP
		Septic leaching rate, per capita	USEPA, 2002
	Food Waste	40% of total available food	Hall et al., 2009
	Manure Application	Animal populations (other than dairy cow)	USDA-NASS, 2017 (2012 data); WSDA (2018)
		Animal excretion rates	Bittman et al., 2019 ; Hong et al., 2011, 2013 ; Nennich et al., 2005; Sheppard et al., 2011
	Crop to Animal Feed	Feed crop production rate	Local agriculture experts (personal communication: WCD); USDA-NASS, 2017 (2012 data)
		Crop N content	USDA-NRCS (2019); local agriculture experts (personal communication: WCD)

Note: Most of the budget results for Canada-NRW were extracted from an existing nutrient budget model that conducted a N assessment for the Lower Fraser Valley in BC (*Bittman et al., 2019*), except for the following: atmospheric deposition, food import, human waste, and food waste. For these fluxes, the same approaches and assumptions were applied to calculate these components for both the US and Canadian portions of the watershed.

Table 2. Nitrogen use efficiencies (NUEs) in the Nooksack River Watershed

	Nitrogen use efficiency (NUE)		
	US-NRW	**Canada-NRW**	**Whole NRW**
Crop NUE (Total Manure and Fertilizer)	54%	22%	51%
Crop NUE (Applied Manure and Fertilizer)	71%	31%	67%
Farm-Gate NUE	19%	45%	27%
Commercial Whole-Farm NUE	24%	53%	33%