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Special Section:
Quantifying Nutrient Budgets for Sustainable Nutrient Management

Key Points:
- Nearly 81% of nitrogen (N) inputs to the Nooksack River Watershed were used to support agricultural production, mostly as animal feed
- About 32% of N export was via ammonia volatilization and 28% via riverine export; 53% of riverine nitrate was from the forested upland
- Contrasts in farm practices and policies between the United States and Canada were reflected in nutrient management, resulting in different NUEs

Supporting Information:
- Supporting Information S1

Correspondence to:
J. Lin,
jlin42@outlook.com

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Author Contributions:
Conceptualization: Jiajia Lin, Jana E. Compton, Shabtai Bittman, David Hooper, Jill S. Baron
Data curation: Jiajia Lin, Chris Clark, Shabtai Bittman, Donna Schwede, Peter S. Homann, Peter Kiffney, Gary Bahr
Formal analysis: Jiajia Lin, Chris Clark, Shabtai Bittman, Donna Schwede, Peter S. Homann
Funding acquisition: Jana E. Compton
(continued)

Key Components and Contrasts in the Nitrogen Budget Across a U.S.-Canadian Transboundary Watershed

Jiajia Lin[1,2], Jana E. Compton[2], Chris Clark[3], Shabtai Bittman[4], Donna Schwede[5], Peter S. Homann[6], Peter Kiffney[7], David Hooper[8], Gary Bahr[9], and Jill S. Baron[10]

[1]Oak Ridge Institute for Science and Education (ORISE), Corvallis, OR, USA, [2]Pacific Ecological Systems Division, U.S. Environmental Protection Agency, Lynden, WA, USA, [3]Whatcom Conservation District, Lynden, WA, USA, [4]Agriculture and Agri-Food Canada, Agassiz, British Columbia, Canada, [5]Center for Environmental Measurement and Modeling, U.S. Environmental Protection Agency, Research Triangle, NC, USA, [6]Department of Environmental Sciences, Western Washington University, Bellingham, WA, USA, [7]National Marine Fisheries Service, Northwest Fisheries Science Center, Fish Ecology Division, National Oceanic and Atmospheric Administration, Seattle, WA, USA, [8]Department of Biology, Western Washington University, Bellingham, WA, USA, [9]Natural Resources Assessment, Washington State Department of Agriculture, Olympia, WA, USA, [10]U.S. Geological Survey, Fort Collins Science Center, Fort Collins, CO, USA

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 (British Columbia, Canada, and Washington, USA) using locally derived data, national statistics, and standard parameters. Feed imports for dairy (mainly in the United States) 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, lower than atmospheric deposition (10%). N imported by returning salmon representing marine-derived nutrients accounted for <0.06% of total N input. 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 United States and poultry products in Canada. Riverine export of N was estimated at 28% of total N export. The commonly used crop nitrogen use efficiency (NUE) metric alone did not provide sufficient information on farming activities but in combination with other criteria such as farm-gate NUE may better represent management efficiency. Agriculture was the primary driver of N inputs to the environment as a result of its regional importance; the N budget information can inform management to minimize N 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 use of reactive 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). Providing the best available information on N sources and transport at different scales can inform 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, multisource, multisectoral N budgets for specific areas of concern. The creation of an N budget is an essential step toward an integrated approach to solving problems associated with excess N release. Input-output budgets can help decision makers better understand and manage N release by quantifying N fluxes at scales appropriate for making management decisions. Many types of accounting approaches have emerged to provide decision makers with information about N sources and loadings (such as NANI, SPARROW, and WSAM) (Hong et al., 2011; Sprague et al., 2000; Swaney et al., 2018). These efforts

Investigation: Jiajia Lin, Jana E. Compton, Shabtai Bittman, Donna Schwede, Peter S. Homann, Peter Kiffney
Methodology: Jiajia Lin, Jana E. Compton, Chris Clark, Shabtai Bittman, Donna Schwede, Peter S. Homann, David Hooper, Jill S. Baron
Project administration: Jana E. Compton
Resources: Jana E. Compton, Chris Clark, Shabtai Bittman, Peter Kiffney, David Hooper, Gary Bahr, Jill S. Baron
Software: Jiajia Lin, Donna Schwede, Peter S. Homann
Supervision: Jana E. Compton, Jill S. Baron
Validation: Jiajia Lin, Jana E. Compton, Chris Clark, Donna Schwede, Peter S. Homann, Peter Kiffney, David Hooper, Gary Bahr, Jill S. Baron
Visualization: Jiajia Lin, Jana E. Compton
Writing - original draft: Jiajia Lin
Writing – review & editing: Jiajia Lin, Jana E. Compton, Shabtai Bittman, Donna Schwede, Peter S. Homann, Peter Kiffney, David Hooper, Gary Bahr, Jill S. Baron

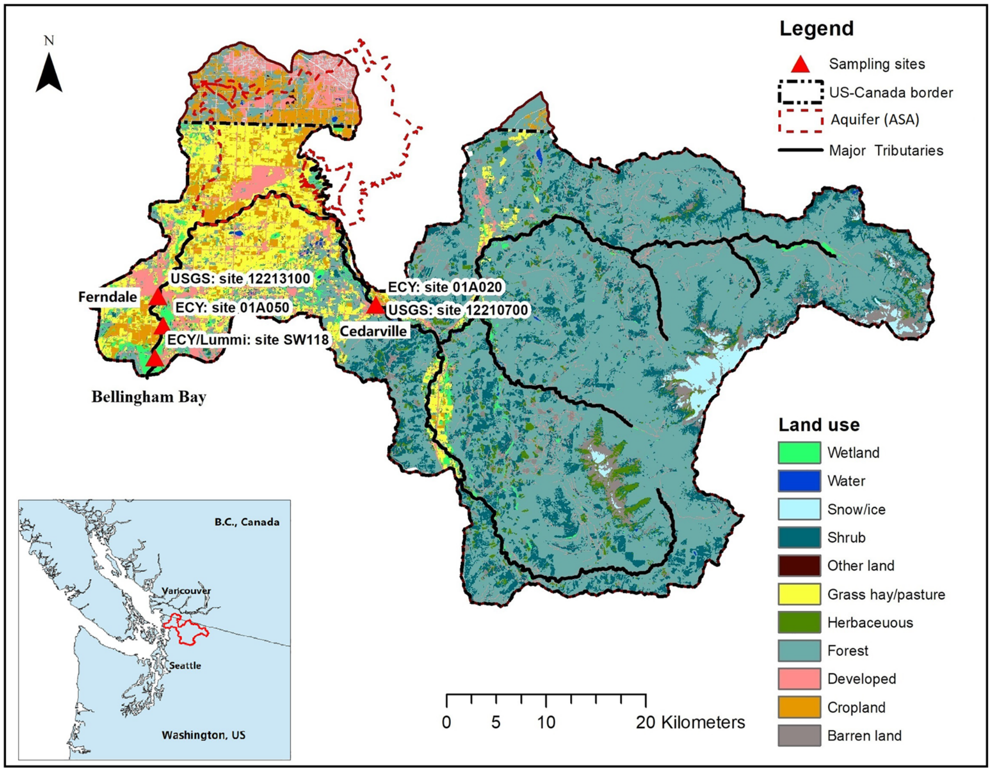

Figure 1. The Nooksack River Watershed (NRW): Nooksack River and its major tributaries and land use. The Abbotsford-Sumas Aquifer (ASA) underlays part of the agricultural land of the NRW in both the United States and Canada. Gaging station measurements and years of data are as follows: U.S. Geological Survey (USGS) Site 12213100: daily discharge (1977–2018) and total Kjeldahl nitrogen (TKN) concentration (1995–1998); Washington Department of Ecology (ECY) Site 01A050: nitrate concentration (1977–2016); ECY/Lummi Site SW118: TKN concentration (2001–2018); USGS Site 12210700: daily discharge (2004–2018); and ECY Site 01A120: nitrate concentration (1977–2016).

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 of the N (Erisman et al., 2003; Galloway et al., 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 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 (WA), USA, and British Columbia (BC), Canada, the Nooksack River Watershed (NRW) supports agriculture, fisheries, wildlife, and urban communities from the North Cascade Mountains to Bellingham Bay in Puget Sound and in BC south of the Fraser River. Agricultural land in the watershed is dominated by berries and forage crops supporting mainly dairy operations, while the important poultry industry imports most of its feed. 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 has elevated 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 ecosystem (Baron et al., 2011; Geiser et al., 2010; Greaver et al., 2012; Sheibley et al., 2014). Elevated N emissions can impair air quality by contributing to particulate matter and ozone precursors that lower visibility and are harmful to human health.

For decades, groundwater nitrate concentrations have exceeded the maximum contaminant level (MCL) for drinking water (10 mg N L^{-1}) in the transboundary Abbotsford-Sumas Aquifer (ASA) (Zebarth et al., 2015). The ASA, which underlies a part of the NRW, is a primary source of drinking water for the transboundary area (Carey et al., 2017) (Figure 1). About 29% of private wells sampled on the U.S. side of the ASA exceeded the MCL (Carey & Cummings, 2013). Recent studies have shown decreasing trends in nitrate concentrations in some 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. However, construction of a transboundary total N budget allows us to integrate information from all sectors, compare different management practices across the border (Robertson et al., 2019), and link these activities to the environmental outcomes. An informal partnership formed in 2016 between scientists and stakeholders in the United States 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 in diverse environments 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 and regional 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. The binational N budget findings could underpin future efforts 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 regional air and water quality.

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 2,130 km^2 area of northwest Washington State in the United States and a smaller portion (6%) in southwestern British Columbia, Canada (Figure 1). Mean annual discharge ranges from 80 to 110 $m^3 s^{-1}$ (Dickerson-Lange & Mitchell, 2014). The watershed climate includes both temperate maritime and Mediterranean 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 (June to September) are generally dry (Pelto, 2015). About 80% of the watershed area lies in mountainous forests dominated by coniferous trees. Urban and residential lands together occupy about 10% of the watershed area, with a total population of over 110,000. The agricultural land area in 2014 was about 174 km^2 on the U.S. side and 42 km^2 on the Canadian side, together comprising 10% of the land area of the watershed. On the U.S. side in 2014, cultivation of forage crops (mainly grass and corn) accounted for about 63% of agricultural land and berries about 21% (Washington State Department of Agriculture [WSDA], 2015). In Canada, berry crops accounted for 80% of cropland. Dairy operations remain an important economic component in the state and in the region despite recent declines in cattle populations; since total dairy cattle population in 1997 was about 40% higher than the current population, there could be a nutrient legacy of organic N in soils that is still contributing N to waters in the watershed (Cox et al., 2018; U.S. Department of Agriculture [USDA], 2017). In 2014, there were about 35,000 dairy cows on the U.S. 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 10 million.

3. Methods

For this study, the U.S. portion of the watershed will be referred to as US-NRW and Canadian portion as Canada-NRW. Due to disparity in available 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 U.S. 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 U.S. and Canadian portions of the watershed.

We integrated data from federal, state and provincial 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 (Figure 2). More details of these methods can be found in the supporting information. We also calculated watershed N retention and use efficiencies. We used 2014 as our target year because of the abundance of monitoring and survey data. When data were not available for 2014, we used data from the closest year (Table 1).

3.1. N Inputs

N inputs to the watershed include atmospheric deposition, food imported for humans and pets, feed imported for farm animals, commercial fertilizer, 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).

3.1.1. 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 4 × 4 km grid resolution. Meteorology was generated using the Weather Research and Forecasting (WRF) model (Skamarock & Klemp, 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 farm emissions from Environment Canada (Bittman et al., 2019). More details of the air quality modeling can be found in Table S1.

3.1.2. Food Import for Humans and Pets

Food consumption by humans was calculated by combining census block population data in both countries (the supporting information), per capita food N consumption values of 4.7 kg N yr^{-1}. The average per capita estimate of U.S. Whatcom County was made using protein nutrition data by age classes (USDA & U.S. Department of Health and Human Services [HHS], 2016). We assumed that all food was imported into the watershed as suggested by local agricultural experts. U.S. population census data for the county and Canadian population and household census data for BC subdivisions were downloaded and clipped to US-NRW and Canada-NRW boundaries, respectively, in ArcMap 10.7 (Environmental Systems Research Institute [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 estimated numbers and nutritional needs of dogs and cats. U.S. Pet Ownership Statistics (American Veterinary Medical Association [AVMA], 2012) showed 37% of U.S. 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).

3.1.3. 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. Cattle populations were estimated based on data from the WSDA dairy inspection program for 2014 (WSDA, 2018). For other animals, we downscaled USDA census data to the Whatcom county level based on the proportion of the county agricultural land 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).

Feed imports were categorized as (1) dairy feed import defined as feed import for lactating dairy cows (population: 20,785); (2) poultry feed import defined as feed import for all poultry such as broilers, layers, turkey,

Table 1
Budget Components and Data Sources (for All U.S. 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	2015 census
		Nutritional consumption, per capita	Hall et al., 2009; USDA & HHS, 2016
	Food import (pet)	Watershed household	2015 census (USDA, 2017)
		Population and body weights (dogs and cats)	Dogs—37% of watershed households; cats—30% of watershed households. Assuming one pet per household; U.S. Pet Ownership Statistics (AVMA, 2012); Baker et al., 2001
		Nutritional and energy needs	Veterinary online manual (Sanderson, 2018); pet basic calorie calculator (OSU Veterinary Medical Center, 2020)
	Feed import	Animal populations (other than dairy cows, such as duck, goat, turkey, hogs, and sheep)	2012 census and survey (USDA, 2017)
		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 (Sanderson, 2018); Bittman et al., 2019; Goyette et al., 2016; Nennich et al., 2005
	Fertilizer import	Crop land	WSDA, 2015
		Crop fertilization rates	Local agriculture experts (personal communication: WCD); Lin et al., 2019; Oregon (Oregon State University [OSU], 2020) and Washington (WSU Extension Publications, 2020) Extension online documentations
	Biological N fixation	Alder density	Ohmann 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
Outputs	Riverine nitrate/TKN export	Flow	USGS Site 12213100 (USGS, 2016)
		Concentrations	Nitrate: WA Department 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, 2017(2012 data); WSDA, 2018
		Synthetic fertilizer application rates	Local agriculture experts (personal communication: WCD); Lin et al., 2019; Oregon (OSU, 2020) and Washington (WSU Extension Publications, 2020) Extension online documentations; WSDA, 2015
		Fertilizer and manure volatilization rate or percentage	Carey & Harrison, 2014; USDA, 1998
	Denitrification loss	Fertilizer and manure denitrification rate or percentage	USDA, 1998
	Animal product (milk)	Dairy cow population	WSDA, 2018 (2014 data)
		Milk N production rate	USDA, 2018; Bittman et al., 2019; Goyette et al., 2016
	Animal product (other)	Animal populations (other than dairy cow)	USDA, 2017 (2012 data)
		Animal product N content	USDA, 2018; Bittman et al., 2019; Goyette et al., 2016
	Crop product	Crop land	WSDA, 2015
		Crop N content	USDA, 2019
	Smolt export	Smolt population and size	Lummi Nation (personal communication: Julie Klacan and Sandra O'Neil, Washington State Department of Fish and Wildlife)
		Smolt body weight equation	Skagit River System Cooperative (personal communication: Eric Beamer, SRSC Research Department)

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Table 1
Continued

	Component	Parameter	Data source
		Smolt body N content	Moore et al., 2011
Internal cycling	Human waste	Sewage treatment plants (STPs) monitored N in effluents	Everson STP (Washington State Department of Ecology, 2020a); Lynden STP (Washington State Department of Ecology, 2020b); Ferndale STP (Washington State Department of Ecology, 2020c)
		Septic population = total population − service population on sewage	USDA, 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, 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
		Immobilization rate	Local agriculture experts (personal communication: WCD)
	Groundwater irrigation	Irrigation amount	Carey & Harrison, 2014
		Groundwater N concentration	Erickson, 1998
	Crop to animal feed	Feed crop production rate	Local agriculture experts (personal communication: WCD); USDA, 2017 (2012 data)
		Crop N content	USDA, 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 nutrient 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 U.S. and Canadian portions of the watershed.

and duck; and (3) feed import for other dairy animals (dry dairy cows, heifers, and calves) and other animals such as beef cattle, goats, equine, and hogs.

3.1.4. Fertilizer Import

A subsurface system has been constructed in parts of agricultural areas in the US-NRW to facilitate manure transportation among farms, which promotes timely and effective manure application in the area. Synthetic fertilizer was imported to supplement manure fertilizer. For US-NRW, we calculated imported synthetic fertilizer as the difference between N "requirement" of each crop and available local manure (as $N_{crop,\ rqr} - Manure_{after\text{-}losses}$). The N "requirement" term described total crop uptake of fertilizer N (both synthetic fertilizer and manure) after accounting for various losses and was calculated as

$$N_{crop,\ rqr} = \sum_{i=1}^{i} A_i \times F_i / f \tag{1}$$

where $N_{crop,\ rqr}$ is the total crop N need (kg N yr^{-1}) in the watershed; A_i and F_i are, respectively, the planting area (ha) (WSDA, 2015) and N uptake (kg N ha^{-1} yr^{-1}) of crop i, suggested by local experts and extension documents (Table S2); and f is a fertilizer coefficient that converts crop N uptake to total applied fertilizer N (both synthetic fertilizer and manure) by factoring various losses such as volatilization and denitrification under local conditions and is calculated as 0.34 for manure and 0.53 for synthetic fertilizer (Table 1; detailed method for f calculation is provided in the 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).

3.1.5. Biological N Fixation

For US-NRW, alder N fixation, a natural N source in the Northwest region (Compton et al., 2003; Wise & Johnson, 2011), 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 estimated because the area grows little N-fixing crops.

3.1.6. 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 (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 U.S. 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

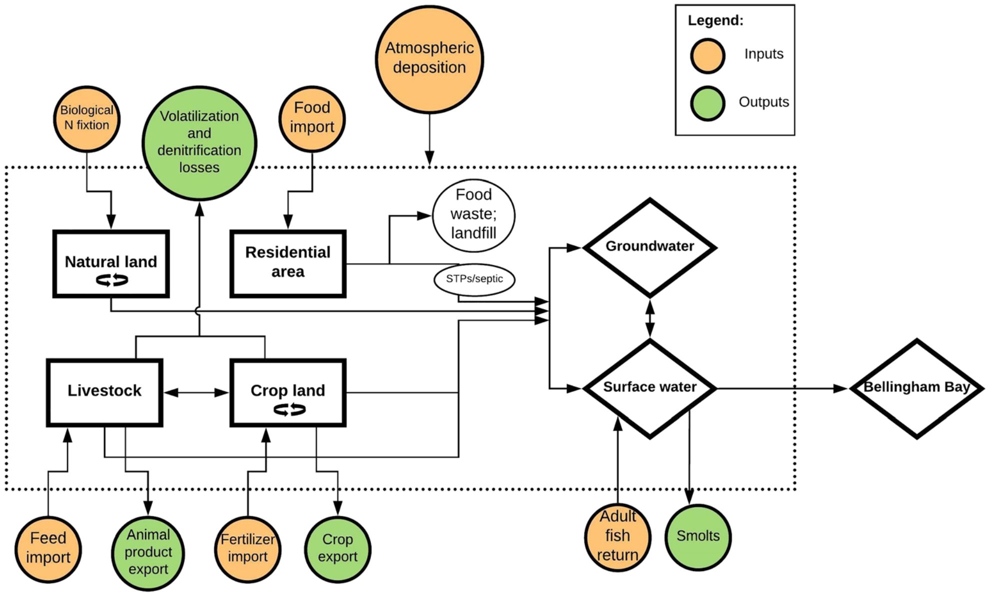

Figure 2. Budget components and N fluxes estimated in the Nooksack River Watershed. Dotted line: watershed boundary. Circles: N inputs (orange) and exports (green) of the watershed; size of the circles is not indicative of the flux magnitude. Squares: watershed land components. Diamonds: hydrologic reservoirs;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

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).

3.2.1. Riverine Export

Total riverine export of N was calculated as the sum of nitrate export and total Kjeldahl N export (TKN, total organic N + total ammonium N). The USGS Load Estimator (LOADEST) model (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%)—this allows us to compare proportional contribution of nitrate from forested versus agricultural land uses; 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. TKN flux was estimated at the downstream site only 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). TKN was not measured at the forested upstream location.

3.2.2. Volatilization and Denitrification Losses

We calculated manure ammonia (NH_3) volatilization in US-NRW based on USDA National Resources Conservation Service (NRCS) estimates for western Washington and information from local agricultural experts (Table 1): We assumed 35% preapplication volatilization loss during manure storage and housing; of the remainder that was applied in field, we assumed an average of 15% volatilization loss for both manure and synthetic fertilizer, which is mostly urea applied by broadcasting (Carey & Harrison, 2014;

USDA, 1998). Ammonia 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 9% of total applied N as manure and synthetic fertilizer in the entire NRW (USDA, 1998). Denitrification in natural lands was not estimated and was assumed to be part of soil N retention.

3.2.3. 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 Equation 2:

$$N_{crop,\,rmv} = \sum_{i=1}^{i} A_i \times Y_i \times (100\% - m_i) \times n_i \qquad (2)$$

where $N_{crop,rmv}$ is the total crop removal of N (kg N yr^{-1}) of the watershed; A_i and Y_i are, respectively, the planting area (ha) and yield (kg crop mass ha^{-1} yr^{-1}) of crop i; m_i is the moisture content (%) of crop i; and n_i is the N content (%) of crop i on a dry weight basis. The 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 products that was very low in N was not calculated.

3.2.4. 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). Nitrogen exported as animal products from Canada-NRW (mainly N in poultry meat and eggs and milk products) was estimated from downscaled data from the Lower Fraser Valley model.

3.2.5. 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 N export was calculated from N content and mass population data. Data and equations were provided by the Skagit River System Cooperative, the Lummi Nation, and the literature (Table 1).

3.3. N Internal Processes

3.3.1. 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 included in 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 (the supporting information). Missing certain sewage treatment plants, total N data were substituted with values 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 mostly from septic tanks and included as river N export from the watershed. To estimate septic inputs to the whole watershed, the population not on sewage was multiplied by an average per capita N excretion rate (4 kg N yr^{-1}) (U.S. Environmental Protection Agency [USEPA], 2002) and assuming a 91% septic leaching rate (USEPA, 2002). In US-NRW, the ratio of the population on sewage and on septic systems was about 2:3. The same ratio was also applied to Canada-NRW.

3.3.2. 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.

3.3.3. Manure Application

Annual manure N application was calculated as 65% of total manure N excreted assuming a 35% preapplication emission loss due to housing, storage, and manure transport. Total animal manure was calculated based on animal populations provided by WSDA and manure excretion rates collected from previous studies

Table 2
Animal Type, Population, and Annual Manure Excretion Rates in the US-NRW

Animal	Population	Manure excretion rate (kg N animal^{-1} yr^{-1}) Low[a]	High[a]	Mean[b]
Lactating cows	20,785	160.60	179.22	169.91
Dry cows	2,987	83.22	83.95	83.59
Heifers (dairy)	5,427	17.55	19.50	18.53
Calves (dairy)	5,783	9.45	9.45	9.45
Broilers	619,773	0.05	0.07	0.06
Layers	5,459	0.50	0.58	0.54
Pullets	330	0.22	0.40	0.31
Turkey	366	0.25	0.89	0.57
Duck	582	0.05	0.06	0.06
Quails	275	0.02	0.08	0.05
Rooster	46	0.50	0.50	0.50
Other poultry	273	0.25	0.25	0.25
Chukars	582	0.06	0.06	0.06
Pheasants	68	0.11	0.11	0.11
Equine	1,308	32.85	54.75	43.80
Alpaca	163	6.95	6.95	6.95
Llama	163	16.92	16.92	16.92
Goats	1,077	6.71	6.71	6.71
Sheep all	800	6.71	6.71	6.71
Beef cow	2,882	43.80	69.35	56.58
Hogs breeder	137	11.68	59.96	35.82
Hogs other	549	5.05	6.52	5.78
Rabbits	52	na	na	na
Cattle on feed	172	43.80	69.54	56.67

[a]The "low" and "high" values are respectively the smallest and greatest rates from previous studies. [b]"Mean" values are the average of "low" and "high" values.

(Table 2). In US-NRW, the total crop N "requirement" and proportional application of manure versus synthetic fertilizer were provided by local farmers for each crop type. Preapplication volatilization loss was taken into account to calculate total manure required. This value was then compared with total 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 feed based on industry data and N in animal products (Bittman et al., 2019).

3.3.4. Crop to Animal Feed

We assumed all the feed crops were consumed in the watershed. 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 imported as soybean, corn grain, and alfalfa and other supplements 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). Most poultry feed was imported, and a small amount was local recycled animal waste.

3.3.5. Soil Processes and Irrigation

Immobilization and mineralization processes in the soil N pool were also considered as internal cycling processes of N. We estimated that after postapplication volatilization loss, about 27% of manure or fertilizer N in the soil was immobilized. A portion of mineralized soil N is taken up by crops, and the rest is leached to groundwater and contributes to elevated nitrate concentrations in groundwater. As there are no estimates or in situ measurements of soil N processes for the NRW, immobilized N was included as N retention in this study.

There could be a substantial amount of groundwater N applied to crops assuming that 10,912 ha of grass and corn received irrigation in the summer. We estimated irrigation N based on previously published groundwater concentrations, irrigation amount, and forage land area: The mean nitrate concentration in 248 wells studied in 1997 (Erickson, 1998) was 5.7 mg N L^{-1}. About 6–14 cm of water was irrigated on forage crops annually in the summer (Carey & Harrison, 2014).

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 estimated 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 (Equation 3). We also calculated NUE for production in the whole watershed using two methods: (1) farm-gate NUE calculated as the ratio of total N exported from farm in products versus total N inputs from all six N sources as described in section 3.1 (Ovens et al., 2008) (Equation 4) and (2) "commercial" whole-farm NUE as the ratio of N in crop and animal products over the import of feed and fertilizer N (Bittman et al., 2016) (Equation 5). All N components in the NUEs calculations were fluxes (t N yr^{-1}).

$$NUE_{crop} = \frac{N_{exp,\,crp}}{N_{app,\,manure} + N_{app,\,syn.frt}} \times 100\% \tag{3}$$

$$NUE_{farm-gate} = \frac{N_{exp,\,animals} + N_{exp,\,dairy} + N_{exp,\,eggs} + N_{exp,\,crp}}{\sum N_{input,\,all}} \times 100\% \tag{4}$$

$$NUE_{commerc.} = \frac{N_{exp,\,animals} + N_{exp,\,dairy} + N_{exp,\,eggs} + N_{exp,\,crp}}{N_{in,\,feed} + N_{in,\,syn.frt}} \times 100\% \tag{5}$$

The NUE for crops helps evaluate the 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. This method 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 farmer's control such as atmospheric deposition and fixation (Bittman et al., 2016; Buckley et al., 2016). It can also help with the assessment of economic consequences.

4. Results

4.1. N Inputs

The combined input of all accounted N sources to the NRW was estimated to be 10,600 t N yr^{-1} (Table S4), equivalent to 50 kg N ha^{-1} yr^{-1} across the entire watershed. N imported as animal feed and synthetic fertilizer was about 8,600 t N yr^{-1} in total and contributed 81% of N influx to the watershed (Figure 3 and Table S4). The largest influx of N was animal feed accounting for about 58% of N inputs to the watershed. Feed imports supporting dairy (milk cows only) 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 U.S. side and much of the poultry production on the Canadian side (Table S4). In the US-NRW, imported feed for dairy cows was more than 3,100 t N yr^{-1}, making up 42% of US-NRW N input. In the Canada-NRW, annual dairy feed import was about 21 t N yr^{-1} representing <0.7% of Canada-NRW N input, while imported feed for poultry was over 2,400 t yr^{-1} representing 78% of Canada-NRW N input. On the watershed level, annual import of over 2,200 t 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 (Figures 3 and 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 N fixation and marine-derived return of adult anadromous fish each represented <0.07% of N inputs. Smaller amounts of these nonagricultural inputs in Canada-NRW than in US-NRW arose because of smaller proportions of land area and total population in Canada, although there also may be a net atmospheric transport of ammonia and NO_2 toward Canada (Figure 3).

4.2. N Exports

The total export of N from the NRW was about 8,500 t N yr^{-1} in 2014 (Table S4) or nearly 40 kg N ha^{-1} yr^{-1} for the entire watershed. The largest N export (32%) was NH_3 volatilization loss (2,745 t N yr^{-1}) (Figure 4 and Tables S3 and S4). Almost 70% of the NH_3 volatilization was associated with dairy manure in US-NRW, with most volatilization (78%) occurring during manure storage and housing processes and 22% occurring after field application (Table S3). Volatilization loss associated with poultry manure was only 11% of total volatilization loss. Export of N in animal products was the second largest flux from the watershed, contributing 2,666 t N yr^{-1} or 31%. Milk was the primary product in US-NRW, and poultry products (meat and eggs) dominated Canada-NRW export (Table S4). 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: 126 t N yr^{-1} as horticulture crops in US-NRW and 64 t N yr^{-1} as berry production in Canada-NRW annually.

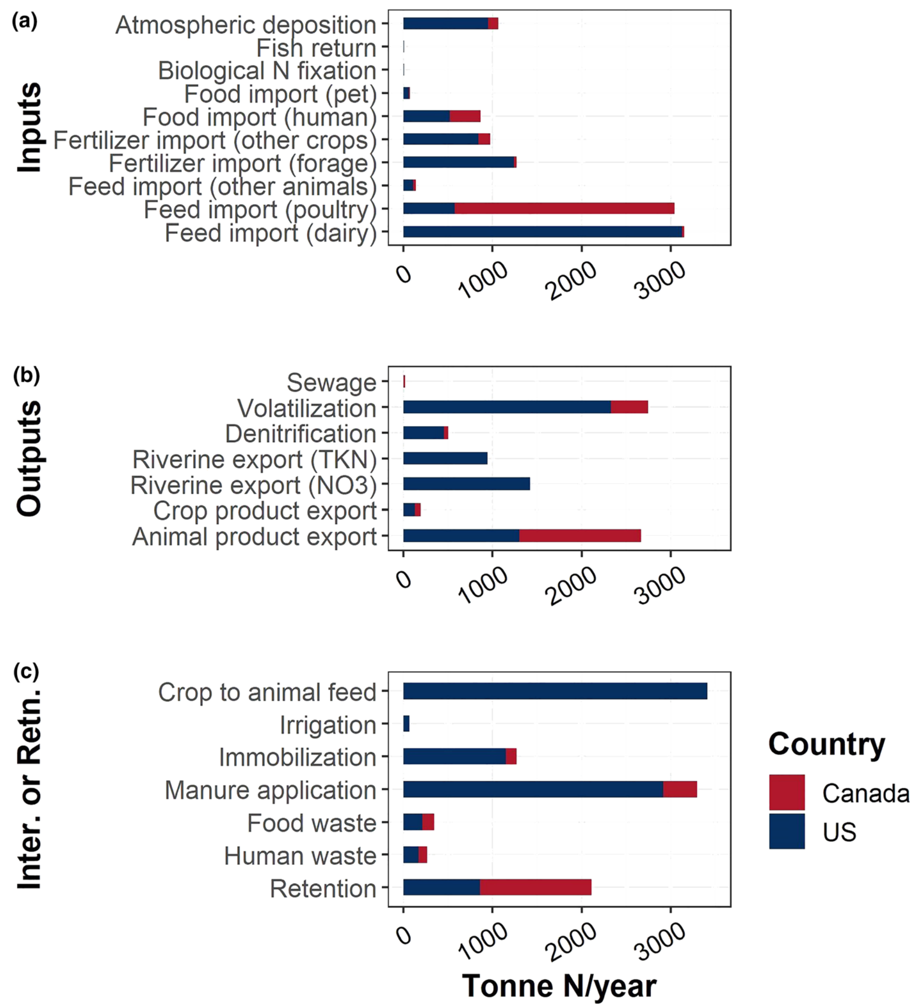

Figure 3. Annual N fluxes of the Nooksack River Watershed (NRW). (a) Inputs. (b) Outputs. "Sewage" refers to N in the effluents from sewage treatment plants in Canada-NRW that drained out of the watershed; export of N via smolt migration is not shown (<0.001%). (c) Internal fluxes or N retention. Irrigation refers to groundwater input to forage crops. "Human waste" refers to N in sewage effluents in US-NRW and septic fluxes in both Canada-NRW and US-NRW. Retention includes immobilization, storage in groundwater, soil, biomass, and unaccounted N losses.

Hydrologic export was 2,160 t N yr^{-1} or 11 kg N ha^{-1} yr^{-1}, accounting for 28% of all N exports or 22% of total N input (Figure 4). In 2014, the Nooksack River transported 1,420 t of NO_3-N and 940 t of TKN into Bellingham Bay (Figure 3), equivalent to an annual export rate of 11 kg TN ha^{-1}. NO_3-N accounted for approximately 60% of the riverine N export. The upland watershed (Cedarville) contributed about 750 t NO_3-N or 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 and thus not shown in Figure 3.

4.3. N Retention, Internal Fluxes, and NUE

The watershed N balance or N retention, calculated as the difference between inputs and exports, was about 2,130 t N or about 20% of total N inputs (Figure 3 and Table S4). This includes losses to groundwater, which we did not quantify. The 20% N retention could be accounted for by accumulation in biomass of perennial crops (e.g., berries) and natural vegetation (e.g., forests), biomass of animals (humans, pets, and stock), and soil retention. It also includes other fluxes such as NO_x emission and natural denitrification such as riparian loss 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 t N to dairy farms in US-NRW to support forage production for nearly 35,000 cows. About

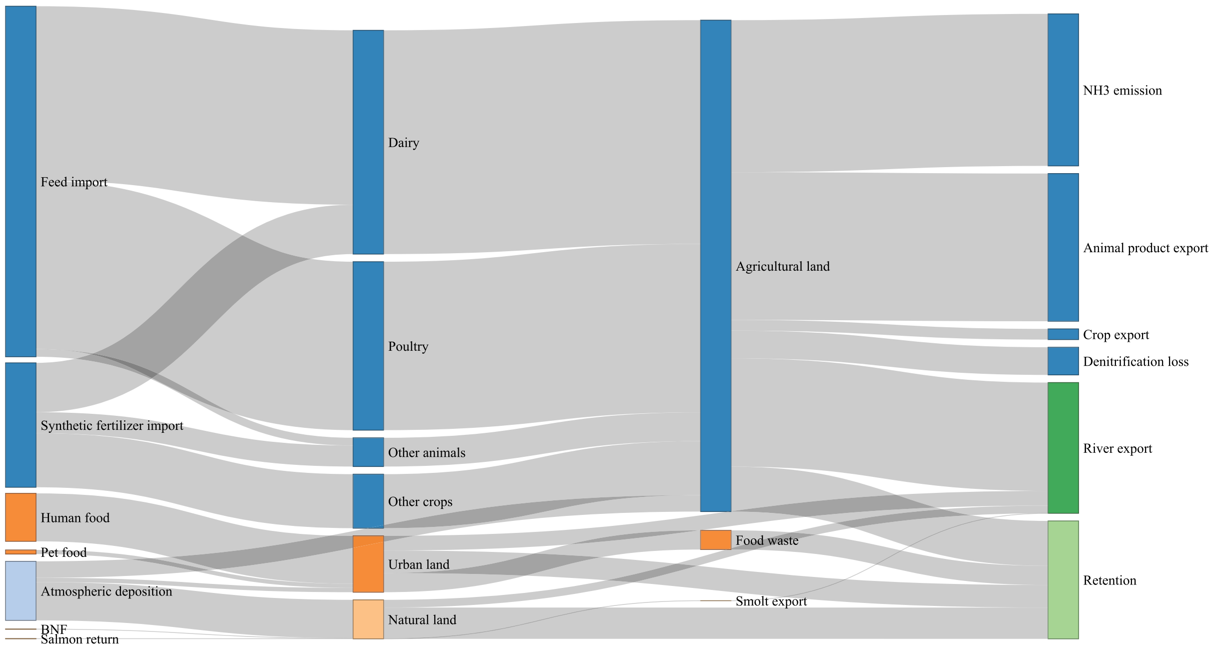

Figure 4. N fluxes in the Nooksack River Watershed. Gray bars represent N fluxes flowing from external inputs on the left side to internal cycling in middle and export/storage on the right. Colors represent sectors: dark blue, agriculture; light blue, deposition; orange, residential; light orange, natural; green, river export and retention within the basin on an annual time step. Bar height is proportional to the magnitude of the N flux.

2,424–2,684 t N from dairy cow manure was applied on the U.S. side. N fluxes associated with dairy cows on the Canadian side were smaller than their U.S. counterparts and much lower than for poultry manure application. In Canada-NRW, local crop feed provided 100 t N for 1,323 dairy cows, and about 96 t N from dairy manure was applied to crops; in comparison, 269 t N in poultry manure was applied to crops (Table S3). We also estimated that the immobilized N from manure and fertilizer applications was about 1,271 t N yr^{-1}. The estimated N loading from irrigation water ranged between 3.6 and 8.4 kg N ha^{-1}, with an average of 6 kg N ha^{-1}. Therefore, the averaged N loading to all forage crops from irrigation water was about 65 t N yr^{-1}. Internal N flux associated with human waste was only 2.4% of inputs or 265 t in the NRW. Unexpectedly, estimated N flux in food waste was 346 t 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 (prevolatilization 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 t). 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 that the crop and animal product export equaled 33% of feed and total fertilizer inputs for the entire watershed, 24% for US-NRW (1,427/5,889 t), and 53% for Canada-NRW (1,433/2,685 t). 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

The N budget identified several key issues related to N management in this transboundary watershed. Agricultural inputs, largely as feed, were responsible for 81% of N inputs to the basin. Exports were dominated by ammonia emissions, animal products, and riverine export (32%, 31%, and 28% of exports, respectively), while 20% of inputs were retained within the basin. The two countries provide contrasts in N management and use efficiencies. These differences are driven by the types of animals raised, manure

Table 3
Nitrogen Use Efficiencies (NUEs) in the Nooksack River Watershed

	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%

management regulations and reporting, and farm economics. Site-specific and feasible nutrient management plans should be developed by accounting for local climate and landscape characteristics and collaboration among land owners and stakeholders. The N budgets and management implications are discussed below.

5.1. Inputs and Internal Cycling

N inputs were high on both the Canadian and U.S. 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 United States (45 kg N ha^{-1} yr^{-1}), but smaller than the Netherlands or China (100 kg N ha^{-1} yr^{-1}) (Liptzin & 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 (the United States) or as fertilizer for forage fed to cattle. Much of the manure produced by animals was applied to crops in both the U.S. 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 and the low N removal by prevalent horticultural crops.

The high N inputs were used with relatively low efficiency (Table 3). The 67% crop NUE for the watershed regarding applied manure and fertilizer was slightly lower than the U.S. national average crop NUE of 70% (Zhang et al., 2015). Crop NUE was higher in US-NRW than in Canada-NRW (Table 3). This could be due to the relative amounts of manure applied. U.S. dairy farmers have ample land for spreading manure for N demanding forages crops, whereas Canadian farmers have limited forage land in this area for spreading manure. Manure transport was not determined due to lack of data. Overall crop NUEs were 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 yr as dairy feed, had N content as high as 3%. Crop NUE alone does not provide comprehensive information on farming activities but could be combined with other criteria such as farm-gate NUE can inform efforts to improve 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 3). 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 & Rotz, 2015): The dairy stocking rate in Canada-NRW was about 2.5 cows per ha. In US-NRW, the dairy stocking rate averaged 3.2 cows per ha for all forage crop land. 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).

Relative to other sources of externally derived N, natural sources (i.e., N fixation in natural lands and salmon returns) were each <1% of inputs at the NRW scale. Returning salmon play a role in supplying organic matter and nutrients to upstream riparian and aquatic food webs (Compton et al., 2006; Gende et al., 2002; Schindler et al., 2003). Although the total mass of N is relatively small, it can have ecological importance. Returning salmon can be significant sources of nutrients and energy to nutrient-poor streams in northern British Columbia and Alaska where salmon populations are much more robust than in the Puget Sound region (e.g., Johnston et al., 2004). Salmon populations in the region are up to 90% smaller than historic levels (Gresh et al., 2000); therefore, we expect that N inputs from spawning salmon were historically higher, especially for nutrient-poor tributaries in the upper reaches of the NRW.

5.2. Release of N to the Environment

Loss of over 50% of N inputs to the environment, primarily as volatilized ammonia and hydrologic 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 changes in biodiversity (Clark et al., 2018; McMurray et al., 2013).

Annual riverine N export from the NRW may contribute to current and future eutrophication and hypoxia in Bellingham Bay (Khangaonkar et al., 2019; Mohamedali et al., 2011). Fractional riverine export for NRW falls within the range of that reported for other watersheds in the Pacific Northwest region (8–30%), such as the Willamette River Basin and Calapooia River Watershed (Compton et al., 2020; Lin et al., 2019). Riverine export rate of the NRW, however, is lower than that of most Willamette and Calapooia subwatersheds and more similar to the export rate of forested subwatersheds and agricultural watersheds in other regions in the United States (Boyer et al., 2002; Schaefer & Alber, 2007; Sigler et al., 2018; Sobota et al., 2009). This is because the NRW has less agricultural land proportionally, different farming practices (less synthetic fertilizer input), and crop types compared to WRB and CRW in spite of their similar climates. Both the forested upland and the agriculturally influenced lowland make substantial contributions to the riverine N export of the NRW. 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 comprised 76% of the whole watershed land area, was >95% forest, and contributed 53% of the riverine NO_3-N export. As a result of forest fragmentation and landscape heterogeneity, forest edges may function as concentrators for wind-borne nutrients (and other pollutants) from adjoining agricultural or urban landscapes (Weathers et al., 2001). Enhanced deposition and throughfall at the edge zone can result in elevated below-canopy chemical fluxes (Pryor et al., 2008; Weathers et al., 2001). These filtering and concentrating functions of edges may help partially explain the upland riverine N export in the NRW. In addition, red alder plays an important in adding fixed N (McCarthy, 2019). Our results support McCarthy's (2019) findings concerning the substantial importance of N inputs from the forested landscape for downstream water quality and nutrient balances in Puget Sound tributaries.

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). Hydrological 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., 2020; De Girolamo et al., 2017; Welter & Fisher, 2016). Wet season precipitation and rising groundwater levels were also linked to high seasonal soil nitrate concentrations, which could lead to elevated N loading to ground waters and high nitrate levels in the aquifer (Carey, 2017; Cox et al., 2018).

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 substantially 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-the-envelope estimation of the range of N flux leaching under different land uses. Based on nitrate leaching rates under a raspberry field (Loo et al., 2019) and a postharvest soil survey of different crops in South Abbotsford and West Sumas (Sullivan & Poon, 2016), townships that are located in northern NRW, we estimated that about 260–430 t N entered the groundwater annually in Canada-NRW, assuming that about 80% of postharvest 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 t N leaching under forage crop land in US-NRW, given that 70% of the forage land there was managed with high intensity. This is equivalent to about 10–13 mg L^{-1} N leaching to groundwater annually under the l0,912 ha of forage land assuming a 0.8 m annual recharge in the US-NRW (Kuipers et al., 2014; Zebarth et al., 1998). 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 estimate that N loss to groundwater could represent about 9–10% of all N inputs or 56–72% of N retention in the NRW.

Much of the applied N could be stored in soil organic matter and remain in the soil for many years to contribute to future crop production, but it may also increase the 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 eventually contributes to surface water export over time, either directly through irrigation using groundwater or indirectly as groundwater flow paths emerge in streams and rivers.

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 might be partially addressed by reducing food waste and dietary N footprints in urban areas (Shibata et al., 2017). 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). To reduce the impacts of excess N, efforts could be promoted through partnerships among the government, community groups, and industry.

5.3. Implications for Effective N Management

Enhancing both dairy and cropping efficiencies is vital to achieving effective nutrient management (Harrison et al., 2007). On the U.S. side of the watershed, there have been many conservation efforts by local and state agencies aimed 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 & Whatcom Conservation District, 2016). Washington State also mandated the development of Nutrient Management Plans for all large commercial dairy farming operations. 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 toward a whole-chain perspective (Sajeev et al., 2018). 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 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, anaerobic digestion, and manure acidification were all found effective in reducing emissions (Sajeev et al., 2018). Adjusting cattle diet such as lowering dietary crude protein was 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; Zhang et al., 2019). 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 strong seasonal hydrology of this area.

Integrated nutrient management may see benefits from reducing imports and seeking export opportunities for excess nutrients. Harrison et al. (2012) suggested that the most effective approach would include accounting for 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 be developed by collaboration with 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; Wassenaar 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 and fertilizer application rates and methods on each farm, which were regarded as confidential business information. This 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, such as wetland and riparian losses.
3. Generalization about certain processes could result in further computational errors. For instance, we used average denitrification (9%) and volatilization (35% preapplication and 15% postapplication) 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% and 30% (Paul & Zebarth, 1997; USDA, 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 1,400 t, 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, 1998), representing a potential error ranging from −68% to 44% in our volatilization estimation.
4. Noncontinuous water sampling and potential errors during sampling and flux simulation (LOADEST) could lead to deviation from the actual riverine N loads. Also, 2014 was a relatively wet year compared to the 20 yr hydrological record. The river discharge in 2014 was about 6% higher than a 3 yr average; therefore, we could have overestimated the long-term hydrologic export of N by only using 1 yr of riverine N result.
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 binational study, resolving issues caused by differences in data collection and resolution between the two countries and the limit of our understanding of the transboundary system 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, exports, 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 into Bellingham Bay was a substantial portion of the export (28%). The agricultural lowland comprised 24% of the entire watershed but contributed 47% of the riverine NO_3-N export. The forested upland watershed comprised 76% of the whole watershed land area and contributed 53% of the riverine NO_3-N export. While the climate and physiography are similar between the United States and Canada in the NRW, the different sides of the border provide contrasts in N management and NUE: Crop NUE was higher on the U.S. 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 United States to Canada (in airflow) and from Canada to the United States (in surface and groundwater flow) will help improve identification of N imbalances and thereby inform strategic policy making to address those challenges.

Data Availability Statement

Repository of data sets generated in this study to calculate nitrogen fluxes is publicly available at EPA repository via the Environmental Dataset Gateway (EDG) (“Transboundary Nooksack River Watershed Nitrogen Budget”; https://www.doi.org/10.23719/1506001) and also available in the supporting information.

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