Nitrogen leaching losses from pasture and winter forage crops in the West Matukituki Valley

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Abstract

In 2013 the Otago Regional Council (ORC) entered mediation with local farmers on changes to their Water Plan, Plan Change 6A, which sought to apply nitrogen load limits within the catchments of Lake Wanaka and Lake Hawea for all farm systems. There was, however, little information available on nitrogen leaching losses in the high rainfall environments (c 2500 mm pa) of the South Island's high country. To address this knowledge gap, a series of monitoring sites were set up in November 2014 in the West Matukituki Valley, west of Lake Wanaka. Porous soil solution samplers were installed to a depth of 50 cm in (i) a sheep- and cattle-grazed pasture, (ii) a cattle-grazed winter forage crop, and (iii) a native bush site. Results for 2015 showed that rainfall totalled 2240 mm, with calculated drainage estimated at 1330 mm. The total fluxes of dissolved N from the crop and pasture sites during the 12 months from June 2015 were 157 and 43 kg N/ha, respectively. Dissolved organic N (DON) made an unexpectedly large contribution to these fluxes, accounting for 20 and 67% of the total fluxes of dissolved N in drainage from the crop and pasture sites, respectively. Gaining a better understanding of the bioavailability of dissolved forms of organic N in leachate would help determine the risks that these forms of nutrients pose to water quality, and if they should be incorporated into models such as Overseer which do not account for DON. The project's initial findings enabled an informed discussion on the use of modelled results from Overseer for the mitigation of N loss at a farm scale in the lakes' catchments. It also focussed farmers' attention on management of the nitrogen-leaky parts of their farm systems, especially winter crops.

Keywords: Inorganic N, dissolved organic N, high rainfall, sheep and cattle grazing

Introduction

The passage of weather systems from west–to–east in Otago and Canterbury mean that heavy rain can occur in the high-country areas of both regions close to the Southern Alps (Macara 2015; 2016). In these alpine valleys rainfall can typically exceed 3000 mm/year. While farms in these environments typically have relatively low stocking rates (Arbuckle and van Reenen 2017), there is still some potential for development (Aspinall 2018). However, farmers are hindered in making decisions on future development by the lack of information on the environmental impacts, particularly N leaching losses, of their current farming practises. With two of the main drivers of N leaching being rainfall (Ledgard et al. 2011) and soil conditions, such as drainage status and texture (Di and Cameron 2002), any further development that increases N cycling in these high rainfall environments would be expected to increase N leaching risk. This is especially true for the valley floors where the soils are predominantly free draining with low nutrient holding capacity (Aspinall 2018). Additionally, the lack of winter pasture growth can result in a reliance on winter forage crops, with the associated increased risk of N leaching under such wintering systems (Shepherd et al. 2012, Smith et al. 2012, Monaghan et al 2013, Smith and Monaghan 2020). While the limited data available does show a link between the amount of pasture cover in the Lake Wanaka catchment and N concentrations in the waterways flowing into the lake (Weaver et al. 2017), there is no data informing how current farming practises are contributing to these losses.

Due to the proximity of these alpine areas to the southern lakes, and the importance of these to New Zealand tourism, communities and regulators are concerned about the effect of farming on lake water quality. In 2013 the Otago Regional Council (ORC) embarked on consultation on their Regional Water Plan change 6A consultation. It was proposed farmers would be restricted to a nitrogen (N) loss of 10 kg/ha/yr within the Lake Wanaka and Hawea Catchments. Along with regulatory change, farmers in these areas are becoming concerned about the interactions between farming practises and community expectations around environmental issues (Aspinall, 2018). The outcome from mediation was for ORC to work with farmers to understand N loss in High Country Environments, supporting an extension project called Farming in a

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Challenging Environment (Arbuckle and van Reenen 2017).

While regulatory bodies in New Zealand, such as ORC, have or are in the process of setting N discharge limits to waters from agriculture, these limits generally relate to losses in the form of nitrite and nitrate-N. Similarly, most of the models used in New Zealand to simulate and predict N losses from agriculture only model inorganic N (nitrate N + ammonium N) (Shepherd and Wheeler 2012). Dissolved organic N (DON) is another form that can also be discharged from pastures yet is commonly overlooked (van Kessel et al. 2009). While there are few reported measurements of DON losses, losses of between 28 – 117 kg N/ha have been reported under pasture (Ghani et al. 2010) and 72-181 kg N/ha under urine patches (Selbie et al. 2015). In summarising DON losses from 16 experimental sites, van Kessel at al. (2009) calculated average DON losses to be 12.7 kg N/ha for all sites and 8.4 kg N/ha when excluding sites representing urine patches. They did note that DON losses increased with increasing precipitation.

This paper reports findings from a N leaching monitoring programme set up as part of the consultation process between ORC and farmers in Central Otago. Measurements of both inorganic and organic N leaching losses were undertaken to give farmers in these environments greater understanding of their N losses.

Materials and Methods

Experimental location and approach

Three locations on Mount Aspiring Station in the West Matukituki Valley (44°30’21”S 168° 46’23”E, west of Lake Wanaka) with contrasting land use were selected: (i) a sheep-cattle grazed pasture, (ii) a cattle-grazed winter forage crop, and (iii) a native bush site. One hundred porous ceramic cups (Webster et al. 1993) for N sampling were installed at both the pasture and forage crop sites in late November 2014. The ceramic cups were placed in two rows of five units within individual blocks (10 m x 10 m in size), with 10 blocks per paddock. The cups were installed at an angle of 45° to a depth of between 45 and 55 cm. An additional 10 Teflon suction cups (MacroRhizon, Rhizosphere Research Products, Wageningen, The Netherlands) were installed at the bush site on the same date. Sampled drainage from these cups was analysed for nitrate and phosphorus (P) concentrations only. Monitoring of drainage at the sites occurred over a 20-month period, from December 2014 until July 2016.

The winter forage crop paddock had been cultivated out of pasture since early October 2014. A swede (*Brassica napus*) and kale (*Brassica oleracea*) crop mix was sown in early November, approximately two weeks before installation of the porous samplers. Lime (1000 kg/ha) was applied to the crop site in October 2014. Fertiliser N and P were applied at sowing at rates equivalent to 49 and 71 kg/ha, respectively. Further applications of 49 kg N/ha occurred in January and February 2015 together with a second application of P (30 kg P/ha) in January 2015. The crop was grazed by cattle (211 calves, 30 R2 cattle and 4 R3 heifers) in late June 2015, then left fallow until early December, at which time it was cultivated and another winter forage crop (kale) sown. The high crop yield meant that the cattle took six days to graze the experimental area of 385 m^2. Three bales of hay each containing 294 kg DM were also fed on the experimental area over this period.

The pasture site was situated in a 16 ha grass paddock adjacent to the crop paddock. It was grazed as part of the normal farm rotation by either sheep or cattle with stock being present most months of the year. Fertiliser inputs consisted of 9 kg N/ha and 12 kg P/ha applied in September 2015. A bush site was also included to check on background fluxes of N below 60 cm. This site was situated across the Matukituki river approximately 1.2 km north of the crop paddock site.

The soil at the pasture and bush sites was predominantly mapped as a Glenorchy Typic Fluvial Recent soil while the crop site was predominantly a Tuamarina Typic Fluvial Recent soil (Hewitt 2010). Key soil features of the three sites were further classified and characterised in January 2015 and are presented in Table 1.

Pasture and crop measurements

Measurements of pasture production were obtained by means of 10 exclusion cages (each 1 m^2) that were placed around the edge of the experimental area. Pasture growth in these cages was measured monthly before they were moved to a new pre-trimmed area. Herbage subsamples from the cages were bulked before being analysed for dry matter (DM) content, species composition and chemical and feed quality values. Herbage chemical analysis was by ICP-OES determination (Khan et al. 2022) following microwave digestion, while feed quality values were by NIRS (Waghorn and Clark 2004).

Crop yield was determined by measuring 4 x 1 m^2 quadrats within the experimental area immediately prior to the grazing that occurred in late June 2015. The crop from each quadrat was weighed, separated into kale or swedes, then dried (60°C) to determine dry matter content. Samples of both forages were then analysed for chemical contents and feed values.

Drainage estimates

A daily soil water balance was used to determine total daily surplus water volumes and soil water deficit

Table 1 Key soil characteristics of the monitoring sites

	Crop site	Pasture site	Bush site
Soil classification	Recent	Recent	Brown
	Young deposits from river	Young fan deposits	Fan deposits with thin loess cover
Texture	Silt to fine sand	Fine to coarse sand	Loamy silt
Depth to gravels	400 mm (range 0 - >600 mm)	220 mm (range 100 - >600 mm)	400 mm (range 260 -500 mm)
Soil water holding capacity to 60 cm depth[1]	163 mm (range 96 - 221 mm)	117 mm (range 84 - 165 mm)	135 mm (range 112 - 161 mm)
Anion Storage Capacity (ASC)			
0-75 mm	16%	13%	42%
0-600 mm	11%	9%	47%
Olsen P[2] (μg/mL)			
0-75 mm	20	16	15

[1]water holding capacity is expressed as a whole soil profile value, corrected for stone content

[2]these values are for the stone-free fraction of the soil

values at each site. The conceptual structure of this balance was adapted from Scotter et al. (1979) and Woodward et al. (2001), with one important difference in that the ratio of actual evapotranspiration to potential evapotranspiration was assessed to have a value of 1.0 when soil moisture contents were between field capacity and a limiting soil water deficit of 50% of plant-available water (Monaghan and Smith 2004). Thereafter, this ratio decreased linearly to become zero at the permanent wilting point. Potential evapotranspiration data were obtained from the Queenstown meteorological station c. 50 km south of the study site. Rainfall data was obtained via telemetry from an ORC weather station located adjacent to the crop and pasture sites. The soil water holding capacity used for this model was taken from the data shown for each site in Table 1 and adjusted for the depth to the porous cups (50 cm).

N and P leaching

Leachate collection occurred when the water balance model indicated that 100 mm of surplus rainfall had been received. Negative pressure (approximately -60 kPa) was applied to sampling ports for at least 18 hours before removing all the collected soil solution. All samples were sent to a commercial laboratory (Eurofins ELS Limited) for chemical analysis by flow injection analysis (APHA 2012). The analytes measured on the samples from the ceramic cups were nitrate + nitrite-N (hereafter referred to as NO_3-N; determined using APHA method 4500 NO_3 or NH_4); ammonium-N (NH_4-N); and total dissolved N (TDN). Inorganic N was calculated as the sum of NO_3-N and NH_4-N, while dissolved organic N (DON) was computed as the difference between TDN and inorganic N concentrations.

Results

Rainfall and drainage

There was 2678 mm of rainfall received at the site between when the measurements commenced (November 2014) and the end of June 2016. There were 24 leachate collections over this period, with an average of 90 mm drainage between each sample collection, although this ranged from 30 to 150 mm depending on rainfall patterns (Figure 1). There was 784 mm of drainage resulting in nine leachate collections before the cows grazed the crop site in late June 2015, and 1325 mm resulting in 15 leachate collections after the crop grazing. For the 12-month period from July 2015 to June 2016 there was 2296 mm of rainfall and between 1343 and 1378 mm of modelled drainage. Despite drainage occurring during most months, events were more concentrated over the winter months and particularly in May and June. The rainfall pattern that occurred during our monitoring period was different to the normal pattern measured at Cascade Creek (8 km W of site) by ORC in that the spring of 2015 was drier than normal and the autumn of 2016 wetter than normal. We therefore evaluated the impacts of contrasting rainfall

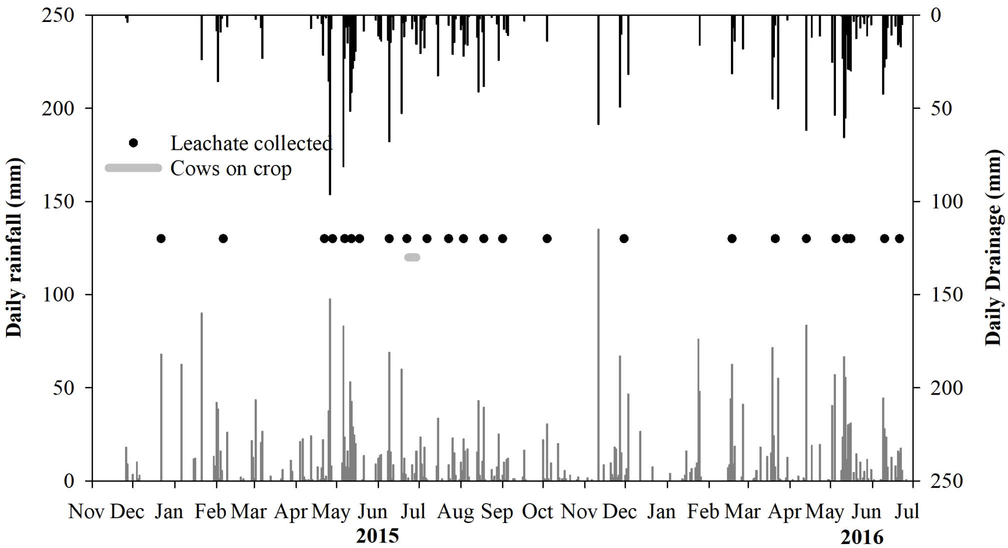

Figure 1 Measured daily rainfall and estimated daily surplus water (assumed to be drainage) at the Matukituki pasture site. The grey bars indicate rainfall and black bars are estimates of drainage.

patterns on drainage and hence N fluxes. This was performed using a probabilistic rainfall and drainage model (Scotter et al. 2000) that was initiated using rainfall distribution patterns and totals measured at the Cascade Creek site from 2004-2016 by ORC. The results of this enabled calculation of more typical annual and seasonal drainage patterns than observed for the 18 months from December 2014 to June 2016.

Pasture and crop production

Pasture production for the 2015 calendar year was 8,251 kg DM/ha. Botanical composition of the pasture was, on average, 74% grass, 7% clover 9% weeds and 10% dead material. The clover content peaked at 13% in February and early December, while the amount of dead material peaked in late autumn and over the winter months.

Chemical analyses indicated that the pasture was consistently deficient in N (range = 2.20 to 3.68%) and deficient in P (< 0.31%), K (<2.27%) and sulphur (S; <0.27%) over the late autumn to winter period. In terms of feed value, pasture metabolisable energy (ME) levels ranged from 8.3 in winter to 10.8 MJ/kg DM in December (normal range = 10 - 11). Crude protein (CP) levels ranged from 14 to 23% (normal range = 10 – 25).

The swede and kale crop produced 19.1 t DM/ha, with an average N content of 1.6%. This meant that the crop contained *c.* 300 kg N/ha before grazing. Crop ME and CP contents averaged 10.6 MJ/kg DM and 13.2%, respectively.

Leachate N concentrations

Nitrate-N concentrations in the crop paddock were initially high, presumably reflecting mineralisation of soil N following cultivation of the paddock (Figure 2A). Concentrations then dropped to background levels of 0.20 - 0.25 mg N/L following 150 mm of drainage, remaining at that level until the winter grazing event. Following grazing of the crop, concentrations then increased steadily to 24 mg N/L after 1500 mm of drainage. The peak post-grazing nitrate-N concentration in the crop paddock was reached in December 2015, although the nitrate-N concentration did not drop to background levels until mid-April 2016. For the 12 months from July 2015 to June 2016, the flow-adjusted mean NO_3-N concentration for the crop site was 7.83 mg N/L (Table 2).

There were no distinct peaks in NO_3-N concentrations in the leachate from the pasture site (Figure 2A). Averaged over the 12 months from June 2015, the flow-adjusted mean NO_3-N concentration for the pasture site was 0.56 mg N/L (Table 2). This contrasts with the bush location, which averaged 0.02 mg N/L.

Ammonium–N concentrations fluctuated at the pasture site and tended to align with grazing pressure (Figure 2B). Average NH_4-N concentrations increased at the crop site immediately following grazing. Mean flow-weighted NH_4-N concentrations were 0.27 mg N/L for the crop paddock and 0.44 mg N-/L for the pasture paddock over the 12-month period from June 2015 to June 2016 (Table 2).

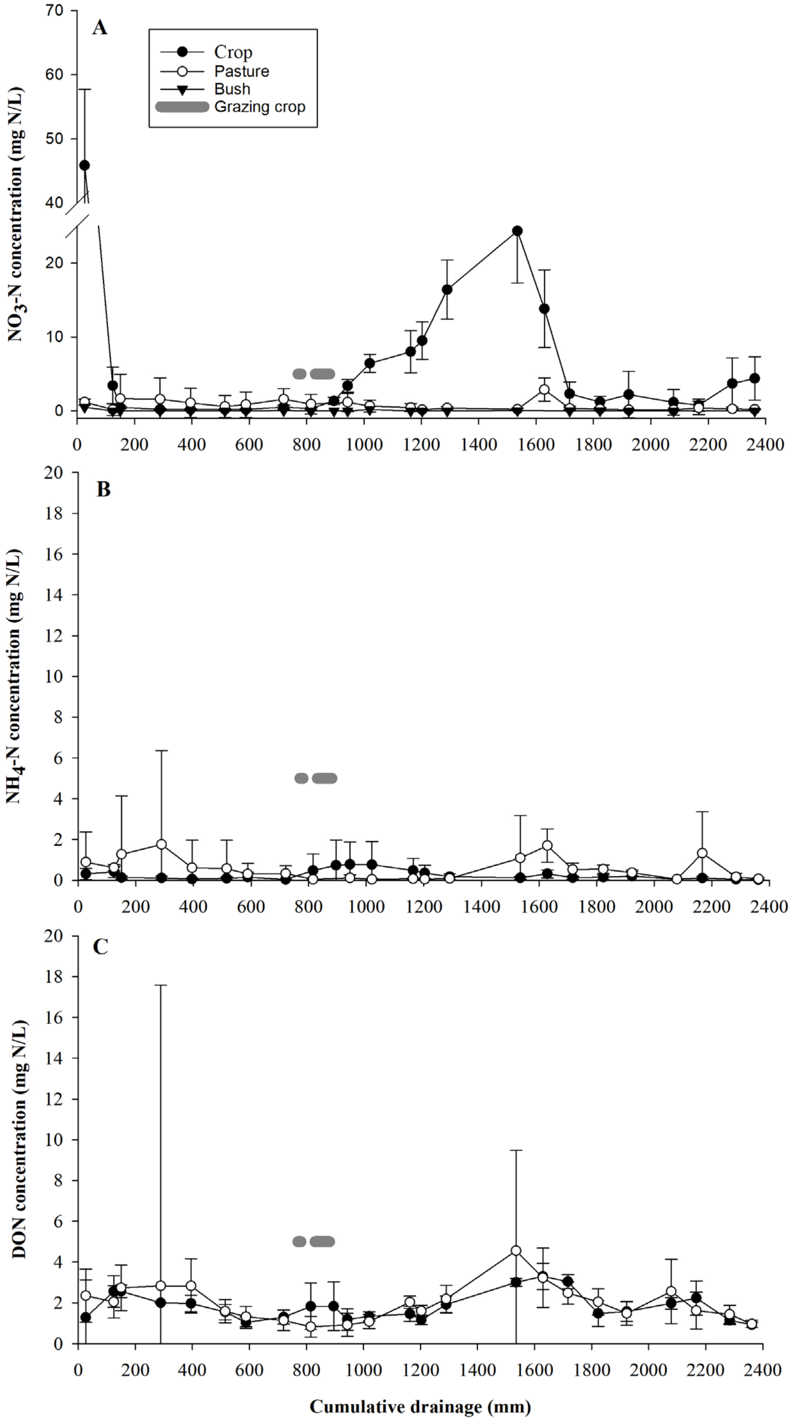

Figure 2 Concentrations of nitrogen (mg N/L) in leachate collected from the Matukituki sampling sites. A) shows NO_3-N for the crop, pasture and bush sites, while B) and C) show NH_4-N and DON respectively for the crop and pasture sites only. Standard Error (SE) terms are shown as bars and the grey dots denote when stock grazed the crop paddock. Note the differing y axes scales.

Table 2 Measured and modelled rainfall and drainage volumes, N fluxes (kg/ha) and mean flow-adjusted N concentrations (mg/L) at the crop and pasture sites in the Matukituki Valley for the period June 2015 to June 2016. "Actual" represents values for modelled drainage based on actual measured rainfall; "Expected" represents values based on a relative rainfall distribution similar to that observed at Cascade Creek (and thus probably more typical of long-term rainfall patterns).

	Crop paddock		Pasture paddock	
	Actual	Expected	Actual	Expected
Rainfall (mm)	2296	2288	2296	2240
Drainage (mm)	1564	1643	1378	1240
Fluxes				
NO_3-N	122	149	8	7
NH_4-N	4	4	6	5
Inorganic N	126	153	14	12
DON	31	30	29	25
Total dissolved N flux	157	183	43	37
Flow-weighted mean concentrations				
NO_3-N	7.83	9.05	0.56	0.58
NH_4-N	0.27	0.27	0.44	0.39
DON	2.00	1.82	2.08	2.02

Concentrations of DON were surprisingly high and greater than measured concentrations of ammonium-N at both the crop and pasture sites (bush site not measured) (Figure 2C). There was little difference in DON concentrations between the crop and pasture sites, except for the December 2015 sampling when there was a considerable increase at the pasture site. Average flow-weighted DON concentrations for the pasture and crop sites were 2.1 mg N/L and 2.0 mg N/L, respectively (Table 2). The patterns of DON elution (Figure 2C) suggest that there was a steady background loss of DON from the crop and pastoral soils, rather than clear pulsed outputs that coincided with excreta deposition.

N leaching losses

Table 2 summarises the estimated N leaching losses and flow-weighted mean concentrations for the crop and pasture sites. Modelled examples are also presented for a scenario ("Expected") based on measured rainfall during the year of study but adjusted for the normal distribution patterns observed for rainfall measured at the Cascade Creek monitoring station. The total losses of dissolved N from the crop and pastures sites during the 12 months from June 2015 were 157 and 43 kg N/ha, respectively. DON made a large contribution to these losses, accounting for 20 and 67% of the total N loss in drainage from the crop and pasture sites, respectively. Drainage losses of dissolved inorganic N for the crop and pasture sites were 126 and 14 kg N/ha, respectively.

Discussion

The measured inorganic N leaching loss of 14 kg N/ha at the pasture site is broadly simiar to the inorganic N losses of 12 to 40 kg N/ha reported in the literature for New Zealand sheep and beef pastures (Betteridge et al. 2007; Hoogendoorn et al. 2011). However, these data come from studies that are generally from areas where rainfall ranged from 700 mm to 1700 mm, thus being considerably lower than the 2500 mm we measured. With rainfall being one of the key drivers of N leaching (Di and Cameron 2002; Ledgard et al. 2011) one would expect the N fluxes under high rainfall conditions to exceed those under lower rainfall conditions. However other factors such as N inputs have an impact on N leaching. Ledgard et al. (2009) showed that nitrate N leaching is closely linked to total N inputs. This is supported by data from Hoogendoorn et al (2016), where on medium slopes with low annual pasture production of just 7.5 t DM/ha inorganic N leaching losses averaged 8 kg N/ha. Our pasture monitoring site with its pasture production of 8.25 t DM/ha, coupled with low inputs of N, would indicate that expected N inputs to the soil via animal excreta would also be relatively low. Thus, despite higher rainfall than other sites, these lower N inputs probably accounted for the relatively low N leaching observed at the pasture site.

Leaching losses of inorganic N from winter grazed crops are known to be higher than those for equivalent pasture sites (Smith and Monaghan 2020). High rainfall at the Matukituki site would suggest that annual N leaching losses would be much greater than reported elsewhere (average of 77 kg N/ha; Shepherd et al. 2012, Smith et al. 2012, Monaghan et al 2013, Smith and Monaghan 2020). The 126 kg N/ha leached from the winter grazed crop that we monitored was indeed greater than this average annual loss but was within the previously reported range of annual N losses (34 – 173 kg N/ha). The low N content of the grazed winter crop is likely to be a factor in this as it would have resulted in lower N intakes and hence excretal N returns than a similar crop with a higher N content (Smith and Monaghan, 2020). The elution curve (Figure 2) indicates that it only took c 1000 mm of drainage to ensure the winter-deposited inorganic N was leached below 55 cm monitoring depth. Thus, one could expect that any excreta N deposited on winter grazed crops in this environment would be leached through the profile in one year and not carried over as has been observed elsewhere (Smith et al. 2012; Smith and Monaghan 2020).

The very low concentrations of nitrate-N found at the native bush site (Figure 2A) are indicative of the low fluxes expected for natural and un-fertilised vegetation. The amount of N transported below 55 cm soil depth represented <3% of the N loss observed under pasture. These small amounts cannot be considered as losses given the potential for uptake by the deeper rooting trees.

The flow-weighted mean concentration of 2.08 mg DON/L measured in drainage from the pasture site was at the upper end of concentration ranges reported for a small number of studies reported in the literature where DON concentrations have been measured under grazed grasslands (0.9 to 2.4 mg DON/L: Watson et al. 2000; Hawkins et al. 2006; Necpalova et al. 2012; Monaghan et al. 2016). The lack of difference in DON concentrations between the pasture and crop paddocks has been noted elsewhere (Monaghan and Smith 2024), suggesting a steady background leaching loss of DON. Combined with a relatively large drainage volume estimated for the Matukituki site, this DON represents a relatively large drainage flux of N compared to losses reported for grazed grassland elsewhere (1.1 to 4.8 kg DON/ha/yr: Watson et al. 2000; Monaghan et al. 2016). The DON flux calculated for the crop site was very similar to the mean DON flux of 22 kg N/ha/yr reported by Smith et al. (2012). However, it must be noted that while DON amounted for 20% of the total N loss under the crop paddock it amounted to 67% of the total N loss under the pasture paddock. While the ecological significance of the large DON fluxes measured in drainage from the pasture and crops is uncertain, Weaver et al. (2017) did find that DON made up 50% of the TN in Lake Wanaka and 59.3% in the Matukituki River. Based on a preliminary study by Ghani et al. (2007), much of the DON in subsurface drainage appears to be readily decomposable and thus potentially bioavailable.

Conclusions

This study has provided quantitative information about N fluxes in drainage from pasture and forage crop sites under high rainfall conditions. The relatively large fluxes of DON highlights the need for gaining a better understanding of the bioavailability and ecological relevance of dissolved forms of organic N in drainage. These findings informed a follow-on extension project with Beef + Lamb NZ, working with Aspiring Environmental, which focussed farmers in the local catchment on management of winter crops and understanding the role of models in catchment management. The Wanaka Catchment Group was formed out of this extension work.

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