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Nitrogen Leaching Losses from Forests in New Zealand

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EXECUTIVE SUMMARY

Regional governing organisations in New Zealand are developing new policies to assess and manage the effects of land use intensification on water quality. Such policies require an assessment to be made of potential N leaching losses from different land uses, particularly under intensification.

This review compiles the information on N concentrations and fluxes in stream and soil water drainage from indigenous and plantation forests in New Zealand. The review showed:

- Nitrogen concentrations in streams draining native forests are similar to those found in undisturbed South American forests minimally impacted by atmospheric pollution. Nitrogen concentrations are approximately ten-fold lower in New Zealand native forests than values estimated for minimally disturbed watersheds in the United States and Finland.

- In common with undisturbed South American forests, dissolved organic N concentrations in native forest streams mostly exceed mineral N concentrations.

- Nitrogen concentrations and yields are higher in streams from native forest influenced by atmospheric inputs from nearby animal agriculture or volcanic vents than in streams not subjected to such inputs.

- Nitrogen concentrations in streams from exotic conifer plantations established on agriculturally unimproved land are generally similar to concentrations from native forest, however limited stream water data are available. Root zone soil water nitrate concentrations of plantations on unimproved land are similar to concentrations in streams draining native forest, but are higher where shrub legumes are present in plantations or for a short period after harvest.

- Nitrogen concentrations in streams draining forests planted on improved pasture land are commonly higher than in streams from native forests, reflecting the prior pasture land-use history. Total N yields from ex-pasture forests were on average more than four times higher than from native forests. However, most of the values for plantations on improved pasture land were from Puruki catchment within the Purukohukohu Experimental Basin, and were influenced by high volcanic N inputs.

The impact of afforestation of fertile pasture land on N leaching is described, as are the effects of plantation forest management practices including fertiliser application, harvesting and herbicide use. Long term data from one catchment study indicate that afforestation of pasture reduces N leaching up until about mid-rotation, after which N losses increase through to harvest before declining again with establishment of the second rotation. Nitrogen leaching is generally not greatly affected by forest management practices, however N fertiliser application can cause large leaching losses in coastal sand forests.

While N leaching from plantations is less than from agricultural land uses, N leaching losses may occur, particularly in forest planted on fertile pasture land and after fertilisation of coastal sands. Minimising leaching will limit adverse effects on the environment and maintain N in the soil for the benefit of crop nutrition. Measures to reduce N leaching losses in forests include:

- Minimising herbicide cover and duration at planting
- Maximising tree N uptake by correcting growth limiting factors such as disease or nutrient deficiency
- Encouraging rapid site revegetation at harvested sites
- Avoiding harvesting in winter and early spring when leaching losses are greatest.
- Splitting application of fertiliser to coastal sands or other sites prone to high N leaching

In catchments with high N soils, where forests may be planted specifically to reduce N loss to sensitive waterways, these practices could be used to maximise the reduction in N leaching. Additionally, nitrification inhibitors may have a role in reducing leaching between canopy closure and harvest when the forest may begin to lose N by nitrate leaching. Use of fast-growing short rotation tree crops could also be considered as these are harvested at a young age, and re-planted, ensuring a continuing high demand on soil N.

REVIEW

The nitrogen (N) cycle in temperate forests is characterised by an almost closed internal cycle between the vegetation and the pool of N in soil organic matter, where the important processes are above and below ground litter production, decomposition, mineralisation (including nitrification), immobilisation, and plant uptake (Gundersen et al. 2006). Except where losses occur as a result of fire, or removal of N in wood products, losses mainly occur as dissolved N in seepage water in the form of ammonium, nitrate, and dissolved organic N (DON). Nitrite is formed as a step in the nitrification process, but is rapidly oxidised to nitrate in most soils. Losses to the atmosphere may also occur In poorly drained soils or riparian zones where nitrate is denitrified to N_2 or other gaseous forms. Losses in drainage water in pristine systems are almost entirely in the form of organic N (Hedin et al. 1995; Perakis and Hedin 2002), elsewhere nitrate, which is highly mobile in soils, is usually the main form lost through leaching.

Nitrogen leaching losses from forest plantations are typically lower than that of other major land uses in New Zealand (Elliot et al., 2005; Hamilton, 2005; Larned et al., 2004). However, application of N fertiliser and other management practices have the potential to increase leaching losses of nitrate and other forms of N from forests (Binkley et al., 1999, Gundersen et al., 2006). Nitrogen leaching losses into rivers and lakes can cause eutrophication resulting in excessive growth of aquatic weeds and algae, which can reduce fish populations and the recreational value of the water. Further, nitrate contamination of drinking water supplies can cause health risks (Cameron et al 2013).

In New Zealand, regional governing organisations are developing new policies to assess and manage the effects of land use intensification on water quality. Such policies require an assessment to be made of potential N leaching losses from different land uses, particularly where land use may be becoming more intensive. This paper compiles and synthesises the information on N concentrations and fluxes in stream and soil water drainage from indigenous and plantation forests in New Zealand with the objective of providing managers with a knowledge base to allow appropriate land use decisions to be made regarding water quality. The forest waters considered are water draining from the root zone, sampled using gravity or tension lysimeters, and water in small forest streams. Root zone drainage water directly reflects the dissolved N response of the plant–soil interactions in the N cycle, whereas stream water reflects additional N processes such as denitrification in the riparian zone and in-stream N conversion (e.g., Cooper et al 1987, Cooper and Thomsen 1988). In-stream processes mainly consume dissolved N, and thus lower concentrations of N are normally found in stream water than in water draining beneath the root zone (Gundersen et al. 2006).

Nitrogen Leaching from Indigenous Forests

Nitrogen in Stream Water

Streams draining undisturbed indigenous forest (and not influenced by atmospheric N inputs from nearby animal agriculture or volcanic sources) contain uniformly low nitrate concentrations. Nitrate concentrations in streams draining both small (< 5 ha) and larger catchments in undisturbed beech-podocarp-hardwood forest of the West Coast and Nelson regions range between 0.01 and 0.05 mg L^{-1} (Appendix, Table 1, Neary et al. 1978, Mosley and Rowe 1981, Fahey and Jackson 1997). Low concentrations were also reported by Stenzel and Herrmann (1990) for 18 small streams draining indigenous forest and three streams draining manuka shrubland in the north and west of the South Island. More recently, a nationwide survey of 97 first order streams draining undisturbed native forest revealed very low (average = 0.007 mg L^{-1}) nitrate concentrations (McGroddy et al. 2008).

Streams draining indigenous forest with adjacent animal agriculture tend to have higher nitrate concentrations. Stenzel and Hermann (1990) found concentrations of 0.17 to 0.46 mg L^{-1} in three South Island east coast peninsular streams from native forest adjoining farmland, while Quinn and Stroud (2002) reported concentrations 0.1-0.12 mg L^{-1} in two streams draining podocarp/hardwood forest at the Whatawhata Research Centre in the Waikato region. Quinn et al. (1997) had earlier reported lower concentrations (0.015 mg L^{-1}) for five streams from podocarp/hardwood forest in the same area. The lower values may reflect sampling in a restricted period (November) when in-stream removal processes are particularly effective (Quinn and Stroud 2002). A national survey of lowland rivers where native forests were the dominant, but not necessarily the only land use, found mean oxidised N (nitrate + nitrite) concentrations of 0.15 mg L^{-1} suggesting an agricultural influence (Larned et al. 2004). Several studies have found consistently higher concentrations (0.7-0.95 mg L^{-1}) in a stream draining podocarp/hardwood forest in the Purukohukohu experimental catchment area in the Bay of Plenty (Cooper et al. 1987, Cooper and Thomsen 1998, Parfitt 2002 & 2003b). While this forest adjoins farmland, the higher concentrations probably reflect atmospheric inputs of 3-6 kg ha^{-1} yr^{-1} from volcanic sources in the area (Parfitt et al. 2002)

Ammonium and DON concentrations in streams draining native forests range between 0.002 and 0.085 and 0.02 and 0.296 mg L^{-1} respectively (Appendix, Table 1). Values for both were highest in streams influenced by inputs from volcanic or agricultural sources at the Purukohukohu and Whatawhata catchments, respectively (Cooper and Thomsen 1988, Quinn and Stroud 2002). In other streams, ammonium concentrations did not exceed 0.04 mg L^{-1} while DON concentrations did not exceed 0.06 mg L^{-1}.

In the study of McGroddy et al. (2008), where the forests were carefully selected as having no evidence of human disturbance in recent decades (and sites where estimates of N deposition indicated substantial deposition inputs from adjacent animal agriculture (> 2 kg ha^{-1} yr^{-1}) were excluded), stream water N concentrations were found to be dominated by organic forms, with DON contributing substantially more (81.3%) than nitrate-N (12.6%) or ammonium-N (3.9%) forms to total N concentrations. While organic N forms dominated in most streams, inorganic forms dominated in 18 out of 98 watersheds. Other New Zealand studies (of sites without volcanic or animal N inputs) followed a similar pattern, though dominance of DON was less marked. This pattern of N loss is consistent with that found in undisturbed temperate forests in southern South America indicating that dominance of organic over inorganic forms is characteristic of undisturbed temperate forests (Hedin et al. 1995; Perakis and Hedin 2002). Nitrogen concentrations in New Zealand streams were similar to those found in Chile and Argentina with values approximately 10-fold lower than those of minimally disturbed forests in United States and Finland, though similar to undisturbed forest in Oregon, USA.

Total N yields from indigenous forest mostly range between 0.25 and 2.5 kg ha^{-1} yr^{-1} (Appendix, Table 1), however up to 5 kg ha^{-1} yr^{-1} may be lost from forests with high volcanic N inputs (Parfitt et al. 2002, 2003b). McGroddy et al (2008) used a simple water balance model and a regression approach to predict DON yields from undisturbed forests with low atmospheric N inputs. Their estimates indicated losses of <0.2-2.5 kg ha^{-1} yr^{-1} for most of the country, but losses of up to 12 kg ha^{-1} yr^{-1} could occur in small areas to the west of the South Island main divide with very high hydrologic throughput. Because of the predominance of organic N in their samples, total N losses should not greatly exceed these estimates.

Nitrogen in Soil Water

The only information on soil water N concentrations in native forests is for five beech forests in the Canterbury and West Coast regions of the South Island (Davis 1990). Concentrations of nitrate-N in soil water extracted by centrifugation from samples collected at 0.3 – 0.4 m depth ranged between 0.004 and 0.041 mg L^{-1} and averaged 0.018 mg L^{-1}. Although ion concentrations in centrifuged samples may be higher than in lysimeter samples (eg Zabowski 1989), these concentrations are within the range of those found in streams draining native forest (Appendix, Table 1).

Impacts of Harvesting

Studies on the effects of harvesting on the chemistry of streams draining native forest catchments have been undertaken in beech/podocarp/hardwood forest at Big Bush in Nelson and at Maimai on the West Coast. At Big Bush, harvesting using either skidder or hauler logging techniques increased total N yields 10-fold. Yields remained 3-5 times higher in logged than control catchments four years after logging and had not returned to control levels by six years after harvest (Fahey and Jackson 1997). Harvesting increased nitrate and organic N but not ammonium N concentrations. Despite the increase in total N, stream concentrations seldom exceeded 1 mg L^{-1}. At Maimai, logging and slash burning increased nitrate yields by up to four-fold in the first year, and up to 20-fold in the second year (O'Loughlin et al. 1980). Nitrate-N yields increased from about 0.5 kg ha^{-1} yr^{-1} in control catchments to up to 10.4 kg ha^{-1} yr^{-1} in a catchment that had been harvested followed by slash burning. Longer term data from catchments that had been harvested with slash either burnt or not burnt indicated more subdued losses of N, of the order of 1.2-2.7 kg ha^{-1} yr^{-1} (compared to control catchment losses of 0.5 kg ha^{-1} yr^{-1}), over an 8-year period (Rowe and Fahey 1991). Ammonium concentrations were increased by slash-burning, but the increase was short lived, with concentrations returning to pre-treatment levels within about two months of treatment (Mosley and Rowe 1981).

Nitrogen Leaching from Plantations

Nitrogen Losses in Stream Water

Forests Planted on Unimproved Land

Few studies have examined N concentrations in streams draining exotic forests planted on 'native' or agriculturally unimproved sites. The measurements available indicate that N concentrations and total N yields are mostly similar to those from native forests (Appendix, Table 2). The lower range in total N yields of these plantations than native forests (Table 1) is a reflection of the lack of available studies. Stenzel and Herrmann (1990) found mean nitrate-N concentrations in seven South Island streams draining exotic forests of 0.021 mg L^{-1}, similar to their value of 0.018 mg L^{-1} for streams draining native forest not influenced by animal agriculture. A national survey of lowland rivers where exotic forests were the dominant, but not necessarily the only land use, found mean oxidised-N (nitrate + nitrite) and ammonium-N concentrations of 0.15 and 0.07 mg L^{-1} respectively (Larned et al. 2004). These values were significantly lower than pastoral classes but not significantly different from native forests in the same study. A stream draining exotic forest in the Rangitaiki catchment, Bay of Plenty region, where nitrate-N concentrations ranged between 1 and 2.5 mg L^{-1} (Collier and Bowman 2003) is an exception to the generally low N concentrations found in exotic forests planted on agriculturally unimproved land. Atmospheric inputs from volcanic activity may have contributed to the higher than normal nitrate concentrations in this stream (e.g. Parfitt et al. 2002).

Forests Planted on Improved Land

Since the 1960's many plantations have been established on improved pasture land during periods when economic returns from livestock farming have been poor. To improve pasture productivity it has been standard practice for New Zealand farmers to apply phosphate fertiliser to stimulate pasture legume growth and N-fixation. Pine plantations planted into improved pasture are likely to have had higher labile soil N levels than native forests or plantations established on agriculturally unimproved land. Nitrogen concentrations in streams draining these forests are commonly higher than in streams from native forests and from forests planted on unimproved land, reflecting the prior pasture land-use history. Nitrate-N concentrations range between 0.037 and 3.5 mg L^{-1}, and average 1.13 mg L^{-1} (Appendix, Table 2). Nitrate-N concentrations were on average six times those of native forest, while ammonium, DON and total N concentrations were 1.4-1.6 times higher. Total N yields from ex-pasture forests ranged from 0.5 to 28 kg ha^{-1} yr^{-1} (Table 1), and at 7.4 kg ha^{-1} yr^{-1}, were on average 4.4 times higher than from native forests. It should be noted, however, that most of the plantations on improved pasture land values were from the Purukohukohu Experimental Basin, and were consequently influenced by high volcanic N inputs.

Table 1. Range in total N yields of native forest and plantations established on unimproved or agriculturally improved land. Ranges for sheep, cattle and dairy cows from Cameron et al. (2013) are given for comparison.

Land use	Total N yield (kg ha^{-1} yr^{-1})	n
Native forest stream water	0.25-5	9
Plantations on unimproved land, stream water	0.04-0.26	2
Plantations on improved land, stream water	0.48-28	9
Plantations on unimproved land, soil water	0-7.1	14
Plantations on improved land, soil water	<1-28	11
Sheep	6-41	2
Cattle	30-56	4
Dairy cows	25-110	7

Nitrogen concentrations and total N yields in the Puruki catchment of the Purukohkohu Experimental Basin have been measured since the plantation was established and have varied greatly over the life of the plantation (Appendix Table 2). Part of the variation is due to sample point location; some samples were from the base of the stream draining the catchment while others were from springs. Additionally, the data of Parfitt et al. (2002) are for a sub-catchment of the Puruki catchment, rather than the whole catchment. Quinn and Ritter (2003) presented Puruki catchment nitrate concentrations and yields over an extended period, from samples taken at the stream weir, the yield data are summarised in Fig. 1 (data not included in Appendix, Table 2). Nitrate yields declined over the 5 years after conversion from pasture to pine in 1972. Average nitrate yield was almost an order of magnitude lower in the subsequent 'young forest' phase (age 5-13, Fig. 1).

Nitrate yields were higher from the 'mature' pine forest, just before logging, than in the young forest phase. Although there are gaps in the monitoring record, this suggests that nitrate retention by the pine forest is greatest when the pine crop has established and is growing vigorously, but the forest becomes more nitrate-leaky as it matures (Quinn and Ritter 2003). Yields and concentrations in the year of logging and the first year after logging increased to similar levels to those during the pasture-pine conversion phase, but decreased quickly towards levels seen during the 'young forest' phase." Parfitt et al. (2002) showed that this reduction was associated with weed growth and an increase in microbial biomass after harvest which would have removed much of the N in the soil solution in the upper soil layers.

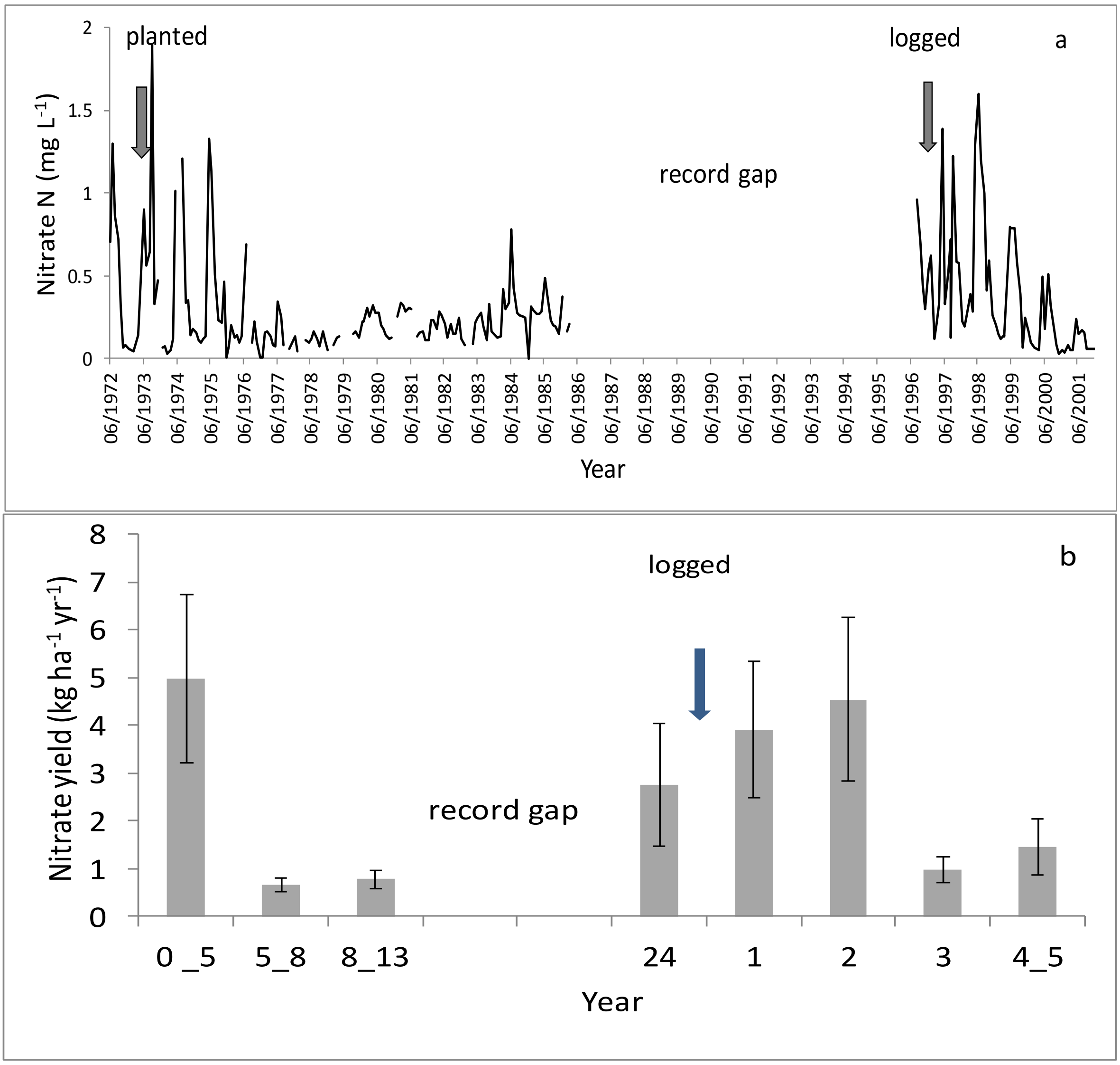

Figure 1. Long term variation in nitrate concentrations (a) and yields (b) in stream flow from Puruki catchment from before pine planting of pasture, through logging to regrowth of the second rotation crop. Bars show standard errors. Data provided by John Quinn, NIWA.

Nitrogen in Soil Water

Forests Planted on Unimproved Land

In forests planted on agriculturally unimproved land, soil water nitrate-N concentrations ranged between 0 and 4 mg L^{-1} (Appendix, Table 3). Highest concentrations were found at Woodhill forest, in samples collected in the first year after harvest of a 32 year old stand, however concentrations at this site rapidly fell to low levels in the following two years (Smith et al. 1994). Elsewhere, concentrations exceeded 0.3 mg L^{-1} at 3 sites, all of which were known to have a history where the N-fixer *Ulex europaeus* was present (Davis et al. 2012) and likely contributed to the higher concentrations at those sites (Dyck et al. 1983, Magesan et al. 2012). Nitrate-N concentrations at other sites did not exceed 0.3 mg L^{-1}, and were mostly less than 0.1 mg L^{-1}, similar to streams draining native forest or plantations on agriculturally unimproved land. Ammonium concentrations ranged between 0.01 and 0.5 mg L^{-1}, and were generally higher at the sites where *U. europaeus* had been present than at other sites. Ammonium concentrations were notably high at the Bulls site, which is coastal sand with low cation exchange capacity and therefor little ability to retain ammonium ions. Dissolved organic N concentrations in soil water generally exceeded mineral N concentrations and exceeded concentrations in streams from both native and plantation forests.

Root zone total N leaching ranged between 0 kg ha^{-1} (as measured by Knight and Will (1977) in 5-12 year old *P. radiata* forest at Kaingaroa Forest - DON not measured), and 7 kg ha^{-1} as measured by Davis et al (2012) at Tarawera in the Bay of Plenty and at Bulls in the Manawatu region (Table 1). Average total N leaching estimated from root zone soil water samples, at 3.25 kg ha^{-1}, was about twice that of streams draining native forest, but about half that estimated for streams draining forests planted on pasture sites, noting again however, that the latter values were dominated by Purukohkohu Basin values.

Forests planted on improved land

The Purukokohu Basin provides most of the data available for root zone leaching from plantations established on improved pasture land. In the Puruki-Rua subcatchment, root zone soil water nitrate concentrations ranged between 0.3 and 3.5 mg L^{-1}, and total N yields ranged between less than 1 and 28 kg ha^{-1} yr^{-1} (Dyck et al. 1987, Parfitt et al. 2002) (Table 1). In a stand thinned from 2000 to 550 stems ha^{-1}, total root zone N leaching increased from mid-rotation (4.9 kg ha^{-1} yr^{-1}) to peak in the year of harvest (28 kg ha^{-1} yr^{-1}), then declined rapidly to low levels (less than 1 kg ha^{-1} yr^{-1}) within two years of re-planting, following the pattern described by Quinn and Ritter (2003) for stream water draining the whole Puruki catchment (see above). A similar temporal pattern is also evident at the Massey farm site where total mineral N yields declined rapidly from 18 kg ha^{-1} yr^{-1} in the year of conversion of pasture to young forest, to less than 1 kg ha^{-1} yr^{-1} in the second year. By year 9, after canopy closure and N demand by the tree crop had declined, leaching had increased to 4.5 kg ha^{-1} yr^{-1} (Parfitt and Ross 2011). A similar transition between ‘non-leaking’ and ‘leaking’ N is apparent at age ten in the second rotation of the Tikitere site of Davis et al. (2012).

Effects of Management Practices on N Leaching in Plantations

Impact of N Fertilisation on Stream Water

Only two studies have examined the effect of N fertilisation on stream water in New Zealand, and both indicate that effects of fertiliser application on stream water chemistry are likely to be minimal. This is consistent with studies reviewed by Binkley et al. (1999) for regions not saturated by atmospheric N deposition. Leonard (1977) studied the impact of fertilisation on stream water draining Pumice soils in Kaingaroa Forest. Nitrogen was applied at 230 kg ha^{-1} as urea to 126 ha immature *P. radiata* stands at varying stages of canopy closure. No attempt was made to avoid the stream channel. The fertiliser was applied to the lower part of the catchment, the upstream part provided an unfertilised control. The total net stream loss due to fertiliser application was 95 kg N (< 1kg ha^{-1}), equivalent to 0.33% of the N applied, half of which was lost during the first week by direct fall into the stream (Table 2). The initial losses were mainly in organic form (as urea) or as ammonia-N, as is typical of fertilisation with urea, particularly if streams are not avoided during fertiliser application (Binkley et al. 1999). The remaining losses were associated with storm events occurring over the next four months. The peak nitrate concentration recorded (1.2 mg L^{-1}) occurred following a rain event 6 weeks after application, while other peaks did not exceed 0.4 mg L^{-1}. Leonard (1977) suggested that avoidance of a strip 20 m wide on either side of the stream would considerably reduce N losses to stream water.

Table 2. Nitrate-N response in stream and root zone soil drainage water to urea fertiliser application.

Soil Group	Fertiliser rate (kgN ha^{-1})	Stand age at fertilisation (years)	Mean NO_3^{-1} conc. unfertilised (mg l^{-1})	Mean NO_3^- conc. fertilised (mg l^{-1})	Increase in mean NO_3^- conc. due to fertiliser (mg l^{-1})	Maximum NO_3^- conc. fertilised (mg l^{-1})[1]	Time after application for maximum (weeks)	Fertiliser N lost to stream water or below root zone[2] (kg ha^{-1})	Lysi-meter depth (m)	Reference
Stream water										
Pumice	200	Immature	< dl[1]	-		1.2	6	< 1 (0.33%)		Leonard (1977)
Brown	92	-	0.32-0.56	-		0.72	2	(<0.1%)		Neary and Leonard (1978)
Brown	92	-	0.19-0.26	-		0.79	7	(<0.1%)		Neary and Leonard (1978)
Root zone soil drainage water										
Pumice	200	12	< dl	< dl	0	< dl	-	0 (0%)	2.7	Worsnop and Will (1980)
Recent	150	2	0-2.5	-	-	24	-	54 (36%)		Thomas and Mead (1992)
Recent[3]	450[7]	0	1.4[8]	8[8]	6.6	18[8]	38	26[9] (6%)	0.6	Smith et al. (1994)
Recent[4]	450[7]	0	2.9	12	9.1	25	98	36 (8%)	0.6	Smith et al. (1994)
Recent[5]	450[7]	0	1.1	8	6.9	24	30	28 (6%)	0.6	Smith et al. (1994)
Recent[6]	450[7]	0	0.8	12	11.2	46	120	45 (10%)	0.6	Smith et al. (1994)
Pumice	200	9	3.14	3.34	0.2	6.15	56	11.5 (5.7%)	1.0	Davis et al. (2012)
Podzol	200	7	0.08	2.79	2.71	3.83	48	28.4 (14.2%)	1.0	Davis et al. (2012)
Recent	200	9	0.06	0.25	0.19	0.53	-	1.7 (0.8%)	1.0	Davis et al. (2012)
Allophanic	200	8	0.03	1.44	1.41	2.54	37	12.1 (6.0%)	0.9	Davis et al. (2012)
Allophanic	200	8	0.04	0.03	-0.01	0.05	-	0 (0%)	1.0	Davis et al. (2012)
Brown	200	8	2.00	37.31	35.31	81.0	29	90.8 (45.4%)	1.0	Davis et al. (2012)
Pallic	200	9	0.32	11.20	10.88	39.3	10	15.2 (7.6%)	0.8	Davis et al. (2012)
Brown	200	7	3.19	2.70	-0.49	2.70	17	0.2 (0.1%)	0.6	Davis et al. (2012)
Brown	200	8	0.05	0.14	0.09	0.43	-	0.2 (0.2%)	1.0	Davis et al. (2012)
Brown	200	8	0.18	3.73	3.55	6.47	21	10.3 (5.1%)	1.0	Davis et al. (2012)

[1] Less than detection limit
[2] Total fertiliser N lost as a % of added N is shown in parenthesis
[3] stem only harvest
[4] stem only harvest, double slash returned
[5] whole tree harvest
[6] whole tree harvest and forest floor removed
[7] 50 kg ha^{-1}, 3-monthly
[8] values estimated from graphs in Smith et al. (1994)
[9] values estimated assuming drainage = rainfall (1330 mm) * 0.3

Neary and Leonard (1978) subsequently monitored total N concentrations in two streams after urea fertilisation (92 kg N ha^{-1}) of forest on central yellow brown earth soils developed on Moutere gravels in Nelson province. One stream channel was overflown while the other was not. Small increases (< 1 mg L^{-1}) in total N concentrations were observed in both streams (Table 2). Overall losses from the two catchments could not be accurately assessed because of lack of stream flow data but the authors estimated that total N losses to stream water amounted to < 0.1 % of that applied.

Impact of N Fertilisation on Soil Water

Increased N concentrations below the root zone following fertilisation indicate a potential for increased stream water concentrations. Studies of the effect of fertilisation on root-zone soil drainage water N concentrations in New Zealand have found average increases ranging from 0 to 35 mg L^{-1} of nitrate -N (Table 2). In a benchmark study, Worsnop and Will (1980) investigated N leaching after fertilisation of recently thinned 13-year-old *P. radiata* growing on pumice soil in Kaingaroa Forest with 200 kg N ha^{-1}. No leaching of N (of any form) was found to 2.7 m depth during the three year monitoring period after fertiliser application. Similarly, fertilisation had no, or minimal, impact on N leaching below 0.6-1 m depth at 6 of 10 sites studied by Davis et al. (2012) where N was also applied at 200 kg ha^{-1} as urea (Table 2), however root zone N concentrations were affected at the remaining four sites. The largest increase in mean nitrate-N concentration (35 mg L^{-1}) occurred on a coastal sand in the Manawatu region. Fertilisation also caused substantial though lower increases in a coastal sand in the Auckland region and on a pallic soil in Canterbury (Table 2). These three sites had in common a recent history where N-fixing shrubs (gorse and tree lupins) were present (Smith et al. 1994, Davis et al. 2012). Factors that seemed to predispose sites to nitrate-N leaching following fertilisation were a recent pasture history, a gorse high component in the understory, and a low soil C/N ratio (Davis et al. 2012).

Maximum nitrate-N concentrations in soil water after fertilisation greatly exceeded those in stream water (Table 2). Denitrification in the riparian zone, in-stream removal and mixing with other water (Binkley et al. 1999, Gundersen et al. 2006) may have all contributed to the smaller effects seen in stream water. Although high peak values were recorded, elevated nitrate-N concentrations in soil drainage water after fertilisation were generally short lived. Average concentrations after fertilisation exceeded 10 mg L^{-1} for at least a year only in the study of Smith et al. (1994) where fertiliser was applied quarterly to a coastal sand at 50 kg N ha^{-1} for 2.5 years. Average concentrations exceeded 10 mg L^{-1} at two sites in the study of Davis et al. (2012), but only for 5-8 months. In contrast, Binkley et al. (1999) found that in a substantial portion of studies they reviewed, average nitrate-N concentrations remained at more than 10 mg L^{-1} for at least a year after fertilisation, possibly reflecting that these sites had reached N saturation from atmospheric N deposition.

In soil water drainage studies, losses below the root zone have ranged from 0-91 kg N ha^{-1} (0-45%) of N applied. The highest losses were recorded on coastal sands in the Manawatu region (91 kg ha^{-1}) and Canterbury (54 kg ha^{-1}) (36 and 45% of N applied, respectively). Lower losses (26-45 kg ha^{-1}, 6-10% of N applied) were recorded in the different harvesting treatments on coastal sand at Woodhill Forest, Auckland (Smith et al. 1994), possibly because N was applied in split applications. However Smith et al. did not measure ammonium-N concentrations and leaching losses may have been underestimated (Thomas and Mead 1992). In an earlier study at Woodhill Forest, Baker et al. (1986) found no difference in N leaching between fertilised (a total of 960 kg ha^{-1} N over ten years) and unfertilised plots. The comparison was made 14 years after the initial N application (not shown in Table 1), and although drainage losses at the time of measurement were low (<0.2 kg ha^{-1} yr^{-1}), significant losses in the past were indicated. For example, in fertiliser treated plots, 340 kg^{-1} N ha^{-1} of the N additions were unaccounted for in biomass (exclusive of roots) and soil N to a depth of 1 m. A proportion of the unaccounted for N would have been contained in root biomass and some may have been lost through volatilisation of ammonia after fertilisation, but most is likely to have been lost through leaching. Losses below the root zone at non-sand sites listed in Table 2 amounted to 0-28 kg ha^{-1} (0-14% of the N applied).

The amount of potentially leachable nitrate-N in soil drainage water arising from fertilisation that would eventually reach streams is not known. Although lysimeters were located below the majority of roots in the studies listed in Table 1, a proportion of tree roots would inevitably occur below the lysimeters at most sites and absorb N. Binkley et al. (1999) knew of no studies that had examined water chemistry profiles of nutrient concentrations through the soil profile into stream water. However Parfitt et al. (2002) found no attenuation of nitrate-N between lysimeters located at 0.6 m depth and spring water in mature *P. radiata* forest on Pumice, suggesting the lysimeter data may accurately simulate the amount of N entering streams. Further studies are required to determine possible N attenuation between the root zone and streams at different sites.

Influence of Harvesting and Subsequent Vegetation Management

Harvesting may lead to increased leaching loss of N from forests as N uptake is disrupted and the soil moisture and temperature climate is changed, however losses appear to be small and short lived, and do not always occur. In a mature *P. radiata* stand in Kaingaroa Forest with low N leaching losses, harvesting marginally increased ammonium-N concentrations in the first year, while harvesting followed by slash burning produced a large increase in surface leachate ammonium-N concentrations, but very little leached below 1 m in any treatment (Dyck et al. 1981). Harvesting caused a rapid increase in nitrate-N concentrations which persisted through to the end of the study (approximately 36 months after logging). Harvesting followed by slash burning caused a shorter lived response. The two treatments caused only minor increases (ca 10 kg ha^{-1}) in the amount of nitrate-N leached over the course of the study (Fig. 2). The plots had been herbicided to control weed growth, which would have exacerbated leaching losses. Parfitt *et al*. (2002) found harvesting reduced catchment nitrate -N loss, from 28 kg ha^{-1} ha^{-1} yr^{-1} prior to harvest to less than 1 kg ha^{-1} ha^{-1} yr^{-1} two years after harvest on a pumice soil of high natural N status at the Purukohukohu experimental basin site. The reduction was attributed to N uptake by weeds (especially grass) that invaded the site after harvest, as well as enhanced microbial activity and incorporation of N into microbial biomass.

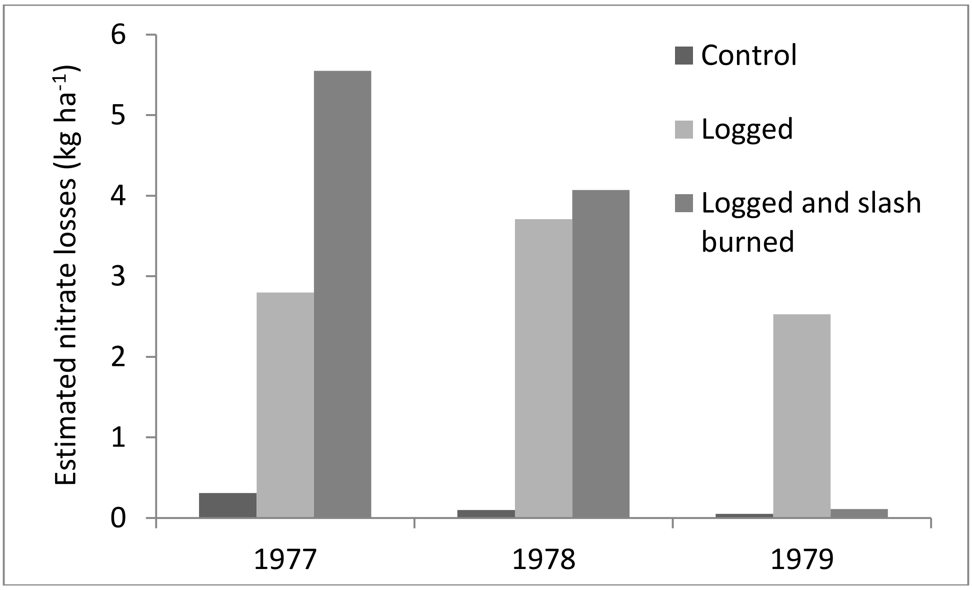

Figure 2. Impact of logging (1976) and burning (1977) operations on estimated nitrate-N leaching losses at 1m depth in *P. radiata* forest on volcanic ash soil, Kaingaroa Forest. Leaching was estimated assuming drainage was one third of annual precipitation (1400 mm). After Dyck *et al*. (1981).

Leaching increases if weed growth after harvest is inhibited. Dyck et al. (1983) removed vegetation from trenched treeless plots (simulating harvesting) in Kaingaroa Forest. After 21 months, soil-water nitrate-N concentrations had risen to a maximum of 9.4 mg L^{-1} compared to 2.0 mg L^{-1} in an adjacent logged but unweeded area. Concentrations in unlogged controls were 0.002 mg L^{-1}. Concentrations declined to 4 mg L^{-1} at the close of the study, 24 months after treatment. Competition removal by herbiciding spots around trees at planting also causes increased leaching losses. Parfitt et al. (2003a) measured N leaching below 0.25 m under herbicided spots, tall pasture between trees, and nearby grazed pasture. The N leaching losses over a 14 month period amounted to 49 kg ha^{-1} under the trees, 15 kg ha^{-1} under the rank grass and 3 kg ha^{-1} under the pasture. The herbicided area under the trees was only 12% of the total land area, so the scaled contribution to leaching under the trees amounted to 5 kg ha^{-1}, compared to 13 kg ha^{-1} under pasture.

Smith et al. (1994) examined the impact of harvest intensity on root-zone nitrate leaching of a second rotation *Pinus radiata* forest on coastal sand dunes at Woodhill Forest in the Auckland region. The harvesting treatments were (1) whole-tree harvest and forest floor removal, (2) whole-tree harvest, (3) stem-only (conventional) harvest and (4) stem-only harvest with double the normal amount of slash returned. The plots were maintained in a weed-free condition throughout the trial. Where trees were conventionally harvested, nitrate-N concentrations at 0.6 m depth increased from about 0.5 mg L^{-1} to 4 mg L^{-1} at 30-40 weeks after harvest before declining to negligible amounts by 80 weeks after harvest. Nitrate concentrations were substantially increased where double the amount of slash was added, but whole tree harvesting and forest floor removal appeared to have little overall effect on nitrate-N concentrations.

Reducing N Leaching from Plantation Forests

While N leaching from plantations is less than from other major land uses, there are key times during the plantation cycle when N leaching may occur, particularly in forest planted on formerly fertile pasture land. Although losses are usually short-lived, minimising N leaching at these key stages will limit adverse effects on the environment and maintain N in the soil for the benefit of crop nutrition.

Loss in Herbicide Spots at Tree Establishment

Herbicide use to reduce competition from weed species is often essential to achieve acceptable tree establishment. Although N losses in individual spots may be relatively high, spots occupy a small proportion of the total land area and so total losses per hectare are not large. Losses in herbicided spots may be kept to a minimum by keeping both the spot area and the duration when the ground is bare of vegetation to a minimum.

Reducing N Loss from Fertilisation

Nitrate leaching losses are usually greatest during winter and early spring when drainage is occurring from the soil and plant uptake of N is low because of cooler conditions, so leaching losses can be minimised by avoiding application at these times. However, if low application rates are used time of application may be less important. Thomas and Mead (1992) investigated measures to reduce leaching losses in coastal sands where losses can be high. Application of 50 kg N ha^{-1} to two-year-old *P. radiata* resulted in minimal leaching losses, irrespective of whether the fertiliser was applied in autumn, spring or summer, with immobilisation in soil accounting for the majority of N not taken up by trees. When a higher rate (150 kg N ha^{-1}) was applied in a single application, up to 30% was lost by leaching below the main root zone (0.3 m) within the first two months, probably as ammonium-N, though little further loss occurred after that (up to 17 months). Splitting the fertiliser into three or nine individual applications resulted in more N being retained in the soil, and no leaching losses of fertiliser N below 0.8 m depth were observed.

Nitrification inhibitors to slow the rate of conversion of ammonium to nitrate in the soil after fertiliser application and in urine patches have been used in pastoral farming in New Zealand to reduce N leaching losses (Cameron et al. 2013), but have not been considered for forest use. Nitrification inhibitors may not work well on coastal sands where forest leaching losses after fertilisation are greatest, as sands have low cation exchange capacity and therefor little ability to retain either ammonium or nitrate. Nitrification inhibitors may also have a limited role at other forest sites found to be prone to N leaching following experimental fertiliser application (forests planted on fertile pasture land or with a high shrub legume content) as, in practice, these sites are unlikely to be fertilised. Nitrification inhibitors may, however, have a role in reducing leaching on N-rich forest sites between canopy closure and harvest when N demand by the crop falls and the forest begins to lose N by leaching (Quinn and Ritter 2003), particularly in catchments where forests may be planted to reduce N leaching to sensitive waterways.

Reducing N Losses at Harvest

Nitrogen leaching losses at harvest may be most effectively reduced by rapid establishment of vegetation cover after harvesting (Dyck et al. 1983, Parfitt et al 2003). A cover of 'weeds' normally develops rapidly after harvest at most forest sites, however where this doesn't occur, grasses or other herbaceous species may be introduced by over-sowing (West et al. 1988, West 1995). Leaching losses can be minimised by avoiding harvesting during winter and early spring.

Maximising Tree N Uptake

Some factor other than N deficiency may be limiting forest growth and preventing maximum N uptake, resulting in increased N leaching. Correction of the limiting factor, for example disease or pest incidence, or nutrient deficiency, may reduce potential N leaching.

Tree Species Selection for Reducing N Losses

European studies have shown that in areas of high atmospheric N deposition, nitrate concentrations in soil drainage water beneath *Picea* (spruce) forests are about double those in drainage water beneath *Fagus* (beech) forests (Rothe 2005). This occurs because *Picea* forests have higher leaf area and, being evergreen, maintain the high leaf area throughout the year in contrast to deciduous *Fagus*. Higher leaf area causes higher interception of atmospheric N and therefore higher N deposition rates. In areas of high atmospheric deposition, planting of deciduous forest species, or species with low leaf area, should reduce N deposition and N leaching. In areas where atmospheric deposition is low (as in most of New Zealand) however, leaching could increase under deciduous or low leaf area species because of reduced canopy interception of precipitation, and greater throughfall and water movement through the soil. The effect of tree species on deposition rates appear to be more important than other species-related processes that might affect N leaching losses (Rothe 2005).

Use of Forests to Reduce N Leaching

Conversion of fertile pasture land to plantation forests rapidly reduces soil N leaching losses (Cooper and Thomsen 1988, Quinn and Ritter 2003, Parfitt et al. 2002) indicating forests have a potential role to play in reducing N leaching losses to rivers, lakes and groundwater. The measures noted above, particularly regarding use of nitrification inhibitors and reducing losses at harvest by ensuring rapid site revegetation, could be used to maximise the 'plantation effect'.

The greatest reduction in leaching occurs early in the life of the plantation during canopy development. After about the time of canopy closure, when the demand for soil N by the tree crop falls, the decline in N leaching induced by the plantation also falls (Fig. 1). In areas where there is a strong need to reduce N leaching caused by intensive agriculture, use of fast-growing short rotation crops such as *Eucalyptus nitens* for wood chip or pulp production could be considered as these are harvested at a young age (10-15 years), and re-planted, ensuring continuing high demand on soil N.

Conversion of pasture to forestry will lead to a reduction in N leaching, however unless the soil N capital available to produce nitrate is significantly reduced, leaching will continue to be a problem. Determination of optimum strategies that reduce the excess N capital in the soil may be aided by use of nutrient balance models such as that developed for *Pinus radiata* (Smaill et al. 2011) which allows determination of the effect of management practices on soil N pools. Modelling should aid in determination of the optimum age of harvest and replanting to achieve maximum reduction in soil N, as well as evaluation of the effects of management practices, such as the removal of harvest residues and litter for off-site use, on soil N capital.

CONCLUSIONS

Nitrogen concentrations in streams draining native forests are similar to those found in undisturbed South American forests minimally impacted by atmospheric pollution. Both mineral N and DON concentrations were approximately ten-fold lower in New Zealand native forests than values estimated for minimally disturbed watersheds in the United States and Finland. In common with undisturbed South American forests, DON concentrations mostly exceed mineral N concentrations.

Nitrogen concentrations and yields are higher in streams from native forest influenced by atmospheric inputs from nearby animal agriculture or volcanic vents than in streams not subjected to such inputs.

Nitrogen concentrations in streams from exotic conifer plantations established on agriculturally unimproved land are generally similar to concentrations from native forest, however limited stream water data are available. Root zone soil water nitrate concentrations of plantations on unimproved land are similar to concentrations in streams draining native forest, but are higher where shrub legumes are present in plantations or for a short period after harvest.

Nitrogen concentrations in streams draining forests planted on improved pasture land are commonly higher than in streams from native forests and from forests planted on unimproved land, reflecting the prior pasture land-use history. Total N yields from ex-pasture forests, at 7.4 kg ha^{-1} yr^{-1} were on average more than four times higher than from native forests. However, most of the values for plantations on improved pasture land were from Puruki catchment within the Purukohukohu Experimental Basin, and were influenced by high volcanic N inputs. Long term data from Puruki catchment show that after fertile pasture is converted to pines, nitrate concentrations and yields decline and remain low until about mid-rotation. Nitrate yields were higher from the ‘mature’ pine forest, just before logging, than in the young forest phase, suggesting that the forest became more ‘nitrate-leaky’ as it matured. Soil water data showed a similar pattern to stream water data.

Only two studies have examined the effect of N fertilisation on stream water in New Zealand plantation forests and both indicate that effects are likely to be minimal, consistent with international studies for regions not saturated by atmospheric N deposition. However fertilisation has been found to increase root zone soil water N leaching at some sites. Factors that seem to predispose sites to nitrate-N leaching following fertilisation are a recent pasture history, the presence of a high component of gorse in the understory, and a low soil C/N ratio.

Harvesting may lead to increased leaching loss of N from forests as N uptake is disrupted and the soil moisture and temperature climate is changed, however N losses appear to be small and short lived, and do not always occur. Leaching increases if weed growth after harvest is inhibited. Competition removal by herbiciding spots around trees at planting also causes increased leaching losses.

While N leaching from plantations is less than from agricultural land uses, N leaching losses may occur, particularly in forest planted on fertile pasture land and after fertilisation of coastal sands. Minimising leaching will limit adverse effects on the environment and maintain N in the soil for the benefit of crop nutrition. Measures to reduce N leaching losses in forests include:

- Minimising herbicide cover and duration at planting
- Maximising tree N uptake by correcting growth limiting factors such as disease or nutrient deficiency
- Encouraging rapid site revegetation at harvested sites
- Avoiding harvesting in winter and early spring when leaching losses are greatest
- Split application of fertiliser to coastal sands or other sites prone to high N leaching

In catchments with high N soils, where forests may be planted specifically to reduce N loss to sensitive waterways, nitrification inhibitors may have a role in reducing leaching between canopy closure and harvest when the forest may begin to lose N by nitrate leaching. In areas where there is a strong need to reduce N leaching caused by intensive agriculture, use of fast-growing short rotation tree crops could be considered as these are harvested at a young age, and re-planted, ensuring a continuing high demand on soil N. Nutrient balance modelling could aid in achieving maximum reductions in soil N capital from different management strategies.

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APPENDICES

Appendix Table 1. Nitrogen concentrations and leaching loss in streams from native forests. Stream water N leaching loss is the sum of the forms shown, unless otherwise indicated in the N leaching loss column.

Forest/ Region	Forest type	Rainfall (mm)	Catchment area and sample period	NO_3-N (mg L^{-1})	NH_4-N (mg L^{-1})	DON (mg L^{-1})	TN (mg L^{-1})	N leaching loss (kg ha^{-1} yr^{-1})	Reference
Maimai West Coast	Beech podocarp hardwood	2610	1.6 ha 12 months (1975)	0.031	0.015	0.061	0.107	1.66	Neary et al. (1978)
Maimai West Coast	Beech podocarp hardwood	1550	1.6 ha 11 months (1979-80)	0.04	0.01				Mosely and Rowe (1981)
Maimai West Coast	Beech podocarp hardwood	1550	2.6 ha 11 months (1979-80)	0.05	0.01				Mosely and Rowe (1981)
Maimai West Coast	Beech podocarp hardwood	1550	1.6, 2.6 ha 9 years (1975-84)	0.05	<0.01			0.40-0.50 (NO_3-N only)	Rowe & Fahey (1991)
Mawh-aeraiti West Coast	Beech podocarp hardwood	1550	406 ha 18 months	0.01	0.01				Mosely and Rowe (1981
Mawh-aeraiti West Coast	Beech podocarp hardwood	1550	56 ha 18 months	0.00	0.01				Mosely and Rowe (1981
Mawh-aeraiti West Coast	Beech podocarp hardwood	1550	71 ha 18 months	0.00	0.01				Mosely and Rowe (1981
Big Bush Nelson	Beech podocarp hardwood	1550	8.6 ha	0.028	0.014	0.042	0.084	0.59	Neary et al. (1978)
Big Bush Nelson	Beech podocarp hardwood	1550	4.8 ha 6 years	0.01	0.04	0.02	0.07	0.29	Fahey and Jackson (1997)
Purukohukohu CNI	Podocarp/ hardwood	1550	28 ha 14 years (1972-1986)	0.805	0.010				Cooper et al. (1987)
Purukohukohu	Podocarp/	1550	28 ha	median 0.836	median 0.010	0.074	0.92	2.93[3]	Cooper &

CNI	hardwood		33 months (1982-1985)						Thomsen (1988)[1]
Purukohukohu CNI	Podocarp/ hardwood	1550	28 ha 33 months (1982-1985)	0.953	0.085	0.296	1.334	0.25[3]	Cooper & Thomsen. (1988)[2]
Purukohukohu CNI	Podocarp/ hardwood	1550	28 ha 4 years (1996-2000)	0.7 (from graph)				5	Parfitt et al. (2002, 2003)
Whatawhata Waikato	Podocarp/ hardwood	1640	52-200 ha 1 month	0.015 NO_3^- + NH_4^+					Quinn et al. (1997)
Whatawhata Native 1Waikato	Podocarp/ hardwood	1720	52 ha 52 months	0.121	0.006	0.122	0.249	2.39[4]	Quinn and Stroud (2002)
Whatawhata Native 2 Waikato	Podocarp/ Hardwood	1720	300 ha 52 months	0.099	0.005	0.092	0.196	2.07	Quinn and Stroud (2002)
Dunlop Creek Westland	Podocarp/ Hardwood (52%), scrub (48%)	-	10 months	0.017	0.002	0.048	0.067		Duggan et al. (2002)
South Island	Indigenous forest	-	18 streams	0.018					Stenzel and Herrmann (1990)
South Island	Manuka shrubland	-	3 streams	0.029					Stenzel and Herrmann (1990)
South Island	Influenced by animal agriculture	-	3 streams	0.286					Stenzel and Hermann (1990)
New Zealand	Indigenous forest	-	97 first order streams	0.007	0.002	0.047	0.056		McGroddy et al. (2008)
NZ lowland rivers	Native forest	-	At least 2 years	0.21	0.07				Larned et al. (2004)

[1] Base flow
[2] Storm flow
[3] Values don't include particulate N
[4] Calculated assuming drainage from pine catchment was similar to that of nearby native forest catchment (962 mm, Quinn and Stroud (2002).

Appendix Table 2. Nitrogen concentrations and leaching loss in streams from plantation forests in New Zealand. Plantations are *Pinus radiata* unless stated otherwise. Stream water N leaching loss is the sum of the forms shown, unless otherwise indicated in the leaching loss column.

Forest/ Region	Forest age and tree stocking	Rainfall (mm)	Catchment area and sample period	NO_3-N (mg l^{-1})	NH_4-N (mg l^{-1})	DON (mg l^{-1})	TN (mg l^{-1})	N leaching loss (kg ha^{-1} yr^{-1})	Reference
Plantations established on agriculturally unimproved land									
Kinleith CNI	Variable aged 1st and 2nd rotation	1340[1]	263 ha 4 months	<0.02	<0.01	0.02-0.30	0.02-0.30	0.26	Leonard (1977)
Tasman Nelson	Not stated	970[1]	> 400 ha 5 weeks				0.32-0.56		Neary & Leonard 1978
Tasman Nelson	Not stated	970[1]	74 ha 5 weeks				0.19-0.26		Neary & Leonard 1978
Taita Wellington	12-14 year old mixed conifers, high gorse content	1295	4 ha 2 years	0.005-0.014	<0.005			0.04 (NO_3^--N only)	McColl et al. (1977)
Rangitaiki BoP	Pre-harvest *P. radiata* & *P. menziesii*	>1300	30-90 ha 1 day	1-2					Collier & Boman (2003)
South Island	6 *Pinus* and 1 *Pinus* + *Eucalyptus* forest	-	1 day	0.021					Stenzel and Herrmann (1990)
NZ lowland rivers	-	-	At least 2 years	0.15	0.07				Larned et al. (2004)
Plantations established on improved pasture									
Whatawhata Waikato	15 year old	1640	5 catchments 43-131 ha 1 month	0.037 (includes NH_4^+-N)					Quinn et al. (1997)
Whatawhata 'pine' Waikato	24-28 year old *P. radiata* (55%) and regenerating scrub	1720	11 ha 4+ years	0.253	0.012	0.154	0.449	4.32[2]	Quinn and Stroud (2002)
Pakuratahi Hawkes Bay	25 year old 225 stems ha^{-1}	1165	345 ha 2 years	0.59	0.02	0.08	0.69	2.29[3]	Fahey & Stansfield (2006)
Puru-kohukohu BoP	0-13 year old 2200 thinned to 160-550 stems ha^{-1}	1550	34 ha 14 years (1972-1986)	0.176 Stream	0.011				Cooper et al. (1987)
Puru-kohukohu BoP	9 year old 550 stems ha^{-1}	1600	34 ha 12 months (1982-1983)	0.60 Spring				0.6	Cooper (1986)
Puru-kohukohu BoP	9-12 year old 160-550 stems ha^{-1}	1580	34 ha 33 months (1982-1985)	0.200	0.016	0.093	0.309	0.59[4]	Cooper & Thomsen (1988)

Puru-kohukohu BoP	9-12 year-old 160-550 stems ha^{-1}	1580	34 ha 33 months (1982-1985)	0.253	0.073	0.246	0.572	0.48[5]	Cooper & Thomsen (1988)
Puru-kohukohu BoP	12 year-old 550 stems ha^{-1}	1550	34 ha 14 months (1985-1986)	Stream				0.7 (NH_4^+ +NO_3^--N)	Dyck et al. (1987)
Puru-kohukohu BoP	24 year old 550 stems ha^{-1}	1817	8.7 ha[6] 12 months (1996)	3.5 Spring Water				28	Parfitt et al. (2002)
Puru-kohukohu BoP	First year after harvest 550 stems ha^{-1}	1589	8.7 ha[6] 12 months (1997)	2.8 Spring				20	Parfitt et al. (2002)
Puru-kohukohu BoP	2nd year after harvest 550 stems ha^{-1}	1863	8.7 ha[6] 12 months (1998)	2.8 Spring				20	Parfitt et al. (2002)

[1] Rainfall estimated by present author
[2] Calculated assuming drainage from pine catchment was similar to that of nearby native forest catchment (962 mm, Quinn and Stroud 2002).
[3] Calculated assuming mean drainage for 1995-1996 = 332 mm (from Wood and Fahey (2006)).
[4] Stream base flow. Values don't include particulate N.
[5] Stream storm flow. Values don't include particulate N.
[6] 'Rua' subcatchment of the Puruki catchment

Appendix Table 3. Soil water N concentrations and estimated N leaching beneath the root zone of plantation forests in New Zealand. All forests are *Pinus radiata* unless stated otherwise. Forests are at least second rotation and are assumed to have been planted on land that has had little or no agricultural improvement. Root zone N leaching is the sum of the forms shown, or as indicated in the root zone leaching column.

Forest/ Region	Forest age, tree stocking	Rainfall (mm)	Lysimeter type[1] depth; sampling period	NO_3-N (mg l^{-1})	NH_4-N (mg l^{-1})	DON (mg l^{-1})	TN (mg l^{-1})	Root zone N leaching (kg ha^{-1} yr^{-1})	Reference
Woodhill Auckland	14 year old 371- 2224 stems ha^{-1}	1330	SC 1 m 12 months	0.026	0.016			0.2	Baker et al. 1986
Woodhill Auckland[2]	1 year old 2500 stems ha^{-1}	1330	SC 0.6 m 12 months	0.5-4					Smith et al 1994
Woodhill Auckland[2]	2 year old 2500 stems ha^{-1}	1330	SC 0.6 m 12 months	0-2					Smith et al 1994
Woodhill Auckland[2]	3 year old 2500 stems ha^{-1}	1330	SC 0.6 m 3 months	0					Smith et al 1994
Kaingaroa CNI	0-3 yr old 2500 stems ha^{-1}	1645	TF 2.7 m 36 months				1.3	4.8 $NH4^{+}$+$NO3^{-}$	Knight and Will 1977
Kaingaroa CNI	5-12 year old 2500→1340 stems ha^{-1}	1500	TF 2.7m 7 years				0	0 $NH4^{+}$+$NO3^{-}$	Knight and Will 1977
Kaingaroa CNI	30-32 year old 321 stems ha^{-1}	1491	TP 1.0m 30 months	0.032	0.011				Dyck et al. 1981
Kaingaroa CNI	20 year old 250 stems ha^{-1}	1500	SC 0.6-0.8 m 24 months	0.006					Dyck et al. 1983
Kaingaroa CNI	56 year old *Pseudotsuga menziesii* 200 stems ha^{-1}	1500	SC 0.6-0.8 m 24 months	0.272					Dyck et al. 1983
Kaingaroa CNI	18 year old *Eucalyptus saligna*	1500	SC 0.6-0.8 m 12 months	0.080					Dyck et al. 1983

	100 stems ha^{-1}								
Shellocks Canterbury	5 year old 1250 stems ha^{-1}	625	TF 0.8 m 24 months					1.33 $NH_4^+ + NO_3^-$ +DON	Watson et al 2003
Doyles Canterbury	22 year old 625 stems ha^{-1}	625	TF 0.8 m 24 months					2.52 $NH4^+ + NO_3^-$ +DON	Watson et al 2003
Mamaku BoP	7 year old 350 stems ha^{-1}	1931	SC 1.0 m 28 months	0.14	0.09	0.62	0.85	6.8	Davis et al 2012
Tarawera BoP	9 year old 250 stems ha^{-1}	2248	SC 1.0 m 24 months	0.05	0.07	0.49	0.51	7.15	Davis et al 2012
Waimarino Wanganui	8 year old 950 stems ha^{-1}	1543	SC 1.0 m 28 months	0.03	0.07	0.63	0.73	5.1	Davis et al 2012
Karioi Wanganui	8 year old 850 stems ha^{-1}	1188	SC 1.0 m 28 months	0.04	0.07	0.61	0.72	2.65	Davis et al 2012
Bulls Manawatu[3]	8 year old 850 stems ha^{-1}	1016	SC 1.0 m 11 months	1.66	0.49	0.48	2.63	7.12	Davis et al 2012
Ashley Canterbury	9 year old 900 stems ha^{-1}	680	SC 0.8 m 22 months	0.31	0.16	0.74	1.21	0.92	Davis et al 2012
Eyrewell Canterbury	7 year old 600 stems ha^{-1}	721	SC 0.9 m 24 months	1.35	0.12	0.95	2.42	2.99	Davis et al 2012
Otago Coast	8 year old 6850 stems ha^{-1} (includes natural regeneration)	833	SC 1.0 m 24 months	0.06	0.06	0.44	0.56	0.35	Davis et al 2012
Catlins Otago	8 year old 550 stems ha^{-1}	1206	SC 1.0 m 27 months	0.13	0.15	0.70	0.98	3.53	Davis et al 2012

[1] Lysimeter type: TF = tension free, T = tension, TP = tension plate, SC = suction cup
[2] Nitrate values were from the 'single slash' treatment of Smith et al. 1994
[3] In Davis et al. (2012), the Bulls site was incorrectly stated as having a land use of fertilized pasture prior to plantation establishment; the pasture is unlikely to have been fertilised (D. Hocking, pers. comm.).

Appendix Table 4. Soil water N concentrations and estimated N leaching beneath the root zone of plantation forests in New Zealand. All forests are *Pinus radiata* and planted on land that was previously fertilised pasture and, except where stated, are first rotation. Root zone N leaching is the sum of the forms shown, or as indicated in the root zone leaching column.

Forest/ Region	Forest age, tree stocking	Rainfall (mm)	Lysimeter type[1] depth; sampling period	NO_3-N (mg l^{-1})	NH_4-N (mg l^{-1})	DON (mg l^{-1})	TN (mg l^{-1})	Root zone N leaching (kg ha^{-1} yr^{-1})	Reference
Puruki BoP	12 yr old 550 stems ha^{-1}	1500	SC 0.6 m 15 months					4.9	Dyck et al 1987
Puruki BoP	12 yr old 2000 stems ha^{-1}	1500	SC 0.6 m 15 months					12.8	Dyck et al 1987
Puruki BoP	23-24 yr old 550 stems ha^{-1}	1817	SC 0.6 m 12 months	3.5				28	Parfitt et al 2002
Puruki BoP	24 yr old harvested April replanted September 550 stems ha^{-1}	1589	SC 0.6 m 12 months	2.8				20	Parfitt et al 2002
Puruki BoP	1 yr old 550 stems ha^{-1} 2nd rotation	1863	SC 0.6 m 12 months	0.28				3	Parfitt et al 2002
Puruki BoP	2 yr old 550 stems ha^{-1} 2nd rotation	1441	SC 0.6 m 12 months					<1	Parfitt et al 2002
Ballantrae Manawatu	20 yr old 250 stems ha^{-1}	1290	T 0.22 m 12 months	0.7				1.9	Parfitt et al 1997
Massey Farm Manawatu	1 yr old 1111 stems ha^{-1}	995	T 0.25 m 12 months					18 NH_4^++NO_3^-	Parfitt et al 2003
Massey Farm Manawatu	2 yr old 1111 stems ha^{-1}	995	T 0.25 m 12 months					<1 NH_4^++NO_3^-	Parfitt et al 2011
Massey Farm	9 yr old 1111 stems ha^{-1}	995	T 0.10 m					4.5 NH_4^++NO_3^-	Parfitt et al 2011

Manawatu Tikitere BoP	9-10 yr old 850 stems ha^{-1} Ex-pasture 2nd rotation	1856	12 months SC 1.0 m 28 months	2.26	0.06	0.51	2.83	21.55	Davis et al 2012

[1] Lysimeter type: TF = tension free, T = tension, TP = tension plate, SC = suction cup