Cultivation of new land: Effects on water quality

A literature review

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Esther Bloem, Marianne Bechmann, Nicholas Clarke and Eva Skarbøvik
Division for Environment

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Cultivation of new land: Effects on water quality - Effekt av nydyrking på vannkvalitet

FORFATTER(E)/AUTHOR(S)

Esther Bloem, Marianne Bechmann, Nicholas Clarke and Eva Skarbøvik

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SAMMENDRAG/SUMMARY:

Denne rapporten er en litteratursammenstilling over tap av suspendert stoff, fosfor og nitrogen fra arealer med hhv. jordbruk og skog/utmark. I tillegg er det gjort en vurdering av tilsvarende tap i perioden der nydyrking gjennomføres. I de norske studiene som er gjennomgått er gjennomsnittlige tap av nitrogen 17 ganger høyere fra jordbruk enn fra skog. Tilsvarende er fosfortap 56 ganger høyere og tap av suspendert stoff 106 ganger høyere fra jordbruk enn fra skog.

This literature survey compares losses of suspended sediments, phosphorus and nitrogen from areas with agriculture and forest/outfields, respectively. In addition, the corresponding losses during the cultivation process are evaluated. In the Norwegian studies referred to here, the average nitrogen losses from agricultural areas are 17 times higher than from forested areas. Correspondingly, the phosphorus losses are 56 times higher and the losses of suspended sediments are 106 times higher from agricultural areas than from forests and outfields.

GODKJENT /APPROVED	PROSJEKTLEDER /PROJECT LEADER
Eva Skarbøvik	Marianne Bechmann
EVA SKARBØVIK	MARIANNE BECHMANN

NIBIO
NORSK INSTITUTT FOR BIOØKONOMI

Preface

This report was prepared with funding from the Norwegian Agriculture Agency and the Nordic Centre of Excellence BIOWATER, funded by Nordforsk under project number 82263. The report provides information to guide water managers in their decision on whether to allow cultivation of new land in catchments with vulnerable water recipients.

Marianne Bechmann and Esther Bloem have collected data from literature and written the main parts of this report. Nicholas Clarke has contributed with literature, text and comments to the section on effects of tree harvest, whereas Eva Skarbøvik initiated the work, and has contributed with literature and text.

Jens Kværner has provided quality control of the report.

In this report losses of nutrients and suspended sediments are presented per decare (daa). Decare is equivalent to 1/10 of a hectare (10 daa = 1 ha).

Ås 10 June 2020

Marianne Bechmann

Project leader

Summary

This literature study compares losses of suspended sediments, phosphorus and nitrogen from areas with agriculture and forest/outfields, respectively. In addition, the corresponding losses during the cultivation process are evaluated. The report is based on literature mainly from Norway and the Nordic countries, but also literature from other countries of similar climate. The method used has been to compare losses of nitrogen, phosphorus and suspended sediments in the following three phases: The original phase – forests or outlying fields; the transition phase – conversion from forest to agricultural land; and the final phase – agricultural land. It is important to emphasise that the method has embedded uncertainty. Existing agricultural land will often be on other soil types than existing forested land. The soil used for cultivation is often in silt- and clay-rich areas, with fertile soils. Most forested areas have thin layers of mineral soils (moraine) over bedrock, or consist of peat soils. The natural background runoff of both nutrients and suspended sediments from such different soil types will therefore be different, but this will depend on the soils in the areas that are cultivated.

Losses of nitrogen from agricultural areas in Norwegian studies were on average 17 times those of areas with forests and outfields. The catchments with extensive livestock (Naurstad and Volbu) were the agricultural catchments with the lowest nitrogen losses, but still on average ten times those of forested areas.

Losses of phosphorus from agricultural areas in Norwegian studies were on average 56 times those of areas with forests and outfields. The catchments with potatoes and vegetables (Vasshaglona and Heia) were among the catchments with the highest phosphorus losses. The maximum phosphorus losses from forests found in this review amounted to 13.9 g P/daa/year, which was nearly three times lower than the minimum measured phosphorus loss in agriculture (41 g P/daa/year).

Losses of suspended sediments from agricultural areas in Norwegian studies were on average 106 times those of areas with forests and outfields.

During the process of new cultivation, special problems are related to tree-harvesting and seedbed preparation. Under various weather conditions, heavy machinery used for forest harvesting of any type may damage the soil and create compaction and the risk for erosion in the wheel tracks. An effect of organic matter is its binding of heavy metals in organic complexes. They may be mobilised together with organic matter and leach into the recipient water.

Changing forests to agricultural land may also affect the hydrology, water temperature, light conditions and therefore affect the composition of biota.

When cultivating new land, environmental mitigation measures should be implemented. Open drainage ditches should be constructed with thresholds to trap sediments, and filter material to hold back dissolved nutrients should be considered. Constructed wetlands and sedimentation ponds should be established at the drainage outlet of the cleared area. A zone with natural vegetation should be left along open channels and streams, and planting of broad-leaved trees should be considered. In the new agricultural land, mitigation measures should be maintained, as described in the Norwegian web-based guidance on measures in agriculture (www.nibio.no).

Contents

1 Introduction

The Norwegian national food policy (St. Prop. 9, 2011-12) has four main goals: Food security, agriculture throughout the country, increased value creation and sustainable agriculture. It is an expressed political target to increase the food production equivalent to population growth. Fulfilling the desire for increased food production can be done by 1) increasing the intensity of food production or 2) increasing the area on which food is grown by cultivating new land.

Cultivation of new land may contribute to more food production and thereby better self-sufficiency for the country, as well as better livelihood for the farmers. Such new cultivation can be carried out in forests, moors and peatlands, but can also replace areas that are taken out of production either for environmental or other reasons e.g. areas set aside for flood land, buffer zones along streams, or land previously occupied by infrastructure. Cultivation of new land implies the removal of trees and branches (called whole tree harvesting; WTH), removal of stumps, and preparing the soil for agricultural land use by drainage and tillage. In intensive forestry, several of these activities are also carried out; among others WTH and stump harvesting. Drainage with open ditches has also been often used in forestry. Mechanical site preparation has been used in some areas. Whole tree harvesting, stump harvesting and soil tillage together represent a major impact on the forest soil. However, whole-tree harvesting is rare and stump harvesting not practiced in Norway. Stem-only harvesting is the normal harvesting method used, but at any one time forest harvesting is only done in a relatively small area as compared to the entire forested area. Once mechanical site preparation has been done, the trees are left to grow for many years, with only minimum management. This is in contrast to agricultural land use systems, which are managed each year. Agricultural land use systems vary largely from extensive grasslands, through cereal production and to intensive potato and vegetable production. Agricultural and forest land use, therefore, cover a wide variety of land use and management practices in addition to natural variation due to location (climate, topography, soil type, etc.). Consequently, the differences in impact of forests and agriculture on water quality will also vary.

Objective

Applications for cultivating new land may be rejected by the environmental departments of the County Governors, since this may have an effect on the water quality of adjacent streams and lakes. According to the Water Framework Directive (2000) and the Norwegian Water Regulation (2007) new interventions shall not reduce the environmental status of water bodies. The Regulation for Cultivation of New Land (2009) states that emphasis should be placed on environmental values when new land is cultivated. An impact assessment may be necessary to approve the clearing of land for cultivation.

In this report, existing knowledge on impacts of cultivation on water quality has been compiled as a basis for the development of guidelines for impact assessments.

This report gives information on the impacts on water quality due to cultivation of new land, by determining impacts of the following three phases:

1. The original phase – forests or other uncultivated areas;
2. The transition phase – conversion from forest to agricultural land;
3. The final phase – agricultural land.

This information can be used to assess the effect of new cultivation by comparing the impact of agricultural land (phase 3) with the impact before cultivation, when the area was forest or outfields (phase 1). In addition, the cultivation process (phase 2) will have activities that can contribute to additional impaired status of water bodies.

2 Methods

This report is based on literature mainly from Norway and the Nordic countries, but also literature from other countries with similar climate is included (see Appendix 1).

Water quality data from *forested* catchments are based on average values from six long-term monitoring sites (Garmo and Skancke, 2018); a literature review by Bækken and Bratli (1995) comparing several monitoring studies at different Nordic forest sites; and a study performed by Vandsemb (2006) on forests and outlying areas (Appendix 1). There is generally more literature on nitrogen than on phosphorus, and the time series are often longer for nitrogen. In the discussion, relevant literature from Norway or Nordic countries was used.

Water quality data from *agricultural* catchments are based on long-term data series from the Norwegian Agricultural Environmental Monitoring Programme (JOVA), together with results from similar Nordic programmes (Stålnacke et al. 2014; Pengerud et al. 2015; Bechmann et al. 2017).

Fewer data exist on the *transition phase,* including the quantification of the effect of whole tree harvesting on stream water quality. However, several studies have measured soil water concentrations of various solutes, including nitrate and to a lesser degree phosphate, before and after harvesting, and some of these including whole tree harvesting. In the absence of streamwater data, the studies of soil solution might be used as a proxy for stream water quality and may illustrate the effect of whole tree harvesting on nutrient status during the transition phase. Literature on the effect of whole tree harvesting on sediment losses is limited.

It is important to stress that the method we have used to assess effects of cultivating forests or outlying fields, has embedded uncertainty. Existing agricultural land investigated by e.g. the JOVA programme, may be on other soil types, geology and climate than existing forested land. The soil that has been used for cultivation for centuries is often located in favourable areas with respect to soil and climate, giving more fertile soils. Most forested areas have more thin layers of mineral soils (moraine) over bedrock, or consist of peat soils. The natural background runoff of both nutrients and suspended sediments from such different soil types and under different climatic conditions will necessarily be different. Hence, direct comparisons of the data on losses of nutrients and sediments from forests/outfields and agriculture, respectively, must be done with care. The extent of uncertainty will also depend on the soil types in the areas that are transformed to cultivated land. Nevertheless, with the lack of data on water quality in rivers and lakes following new cultivation, this was the best approximation that could be achieved within the resources of the project.

3 Sediment and nutrient losses from forests and outfields

Of Norway's total land area (385 200 km²), 70 000 km² is productive forest of which 420 km² is harvested every year. In addition 220 000 km² is outfields. New cultivation may occur on both forests and outfields.

In Table 3.1 a summary of the main findings with respect to nitrogen, phosphorus and suspended sediment losses are presented. These results are discussed in detail in the sections 3.1 Nitrogen loss, 3.2 Phosphorus loss and 3.3 Suspended sediments - erosion.

Table 3.1: Average losses of total nitrogen, total phosphorus and suspended sediments in the studied forest catchments. In brackets the range of average annual losses from the different catchments. (Garmo and Skancke, 2018; De Wit and Wright, 2008a; Vandsemb, 2006; Bækken and Bratli, 1995)

	Nitrogen average kg Tot N / daa /year	Phosphorus average g Tot P / daa / year	Suspended sediments average kg SS / daa / year
Norwegian forests	0.32 (0.07-1.50)	5.07 (0.31-13.90)	0.95 (0.25-2.20)
Nordic forests	0.24 (0.06-1.50)	5.75 (0.02-14.60)	0.76 (0.02-2.20)
Field studies – before harvesting*	0.13 (0.09-0.20)[1]	7.80 (1.90-14.60)[2]	0.40 (0.40-0.40)[3]
Field studies – after harvesting*	0.49 (0.23-0.95)[1] Increase of 3.7 times	28.27 (9.30-65.50)[2] Increase of 3.6 times	0.50 (0.40-0.60)[3] Increase of 1.25 times

* [1] Based on Table 3.2, [2] Based on Table 3.3, [3] Based on Table 3.4.

3.1 Nitrogen losses from forests

Annual losses of total nitrogen (Tot N) from forests and outfields varies in Nordic catchments (Norway, Sweden and Finland) from 0.06 to 1.50 kg Tot N/daa/year (Table 3.1; Figure 3.2; Attachment 1). The average nitrogen losses for these study sites is 0.24 kg N/daa/year. The average nitrogen losses for the Norwegian catchments are higher (0.32 kg N/daa/year) than the Swedish and Finnish catchments (0.17 kg N/daa/year), which may be due to generally higher precipitation and runoff (Bækken and Bratli, 1995).

The highest nitrogen losses from forests and outfields were found in the catchments of Vasshaglona Skog in Agder County, Dal in Vestfold and Telemark Counties, Svela in Rogaland County and Øygardsbekken in Rogaland County (Figure 3.1). In Øygardsbekken, shallow soil, high precipitation and high N-deposition contributed to the high nitrogen losses (Schartau et al., 2006). For the three catchments of Vasshaglona Skog and Dal, years with tree-harvesting were included in the time-series, which may explain high nitrogen losses from these catchments (Vandsemb 2006; Kaste et al., 1996).

Seven study sites in the Nordic countries have been monitored for nitrogen losses before and after harvesting (Table 3.2, Bækken and Bratli, 1995). The average nitrogen losses before harvesting were 0.13 kg N/daa/year and increased to an average of 0.49 kg N/daa/year after harvesting; this is an increase of 3.7 times. A more recent study by De Wit et al. (2014) presented two sites at Langtjern and found nitrogen losses before and after harvesting to be 0.085 kg N/daa/year (2008) and 0.311 kg N/daa/year (2009-2012). Before harvesting, the nitrogen losses at Langtjern were similar to nitrogen losses at another site, Sniptjern (Table 3.2), while after harvesting the nitrogen losses were 45% lower than at Sniptjern. This difference in effect of tree-harvesting may be related to the amount of trees and braches removed, and possible mineralisation of organic matter following tree-harvest. Langtjern has thin mineral soils and low biomass production. Time could also play a role: In a literature review, Gundersen et al. (2006) found that the increase in nitrogen losses returned to pre-harvest levels after 3-5 years.

Table 3.2. Nitrogen losses in Nordic forest areas before and after harvesting (Bækken and Bratli, 1995).

	Period		N kg/daa/year		Source
	Before harvest	After harvest	Before harvest	After harvest	
Norway					
Andebu II	1972-1975	1975-1977	0.14	0.95	Haveraaen, 1981
Sweden					
Kloten	1970-1977	1-3 år	0.106	0.241	Grip, 1982
Kullarna	1977-1980	1-5 år	0.092	0.518	Rosén, 1982, 1987
Sniptjern	1977-1980	1-5 år	0.088	0.567	Rosén, 1982, 1987
Finland					
Murtopuro	1972-1982	1982-1985	0.204	0.43	Ahtiainen, 1992
Kivipuro (med kantsone)	1972-1982	1982-1985	0.157	0.226	Ahtiainen, 1992

3.2 Phosphorus losses from forests

Average losses of total phosphorus (Tot P) from forests vary in Nordic catchments (Norway, Sweden and Finland) from 0.02 to 14.6 g Tot P/daa/year (Table 3.1 and Figure 3.2; Bækken and Bratli, 1995; Vandsemb, 2006; Garmo and Skancke, 2018). The average phosphorus losses for these study sites were 5.75 g P/daa/year. The average phosphorus losses from the Norwegian catchments were slightly lower (5.1 g P/daa/year) as compared to the Swedish and Finnish catchments (average 7.6 g P /daa / year). There are no obvious outliers with particular high phosphorus losses within the dataset (Figure 3.2). One catchment, however, had very low phosphorus losses (Svartbekken, see Appendix 1). This catchment is dominated by non-productive forest.

An increase of phosphorus losses is expected after harvesting (clearcutting). Swedish calculations indicate a doubling of phosphorus losses after forest harvesting. Three Scandinavian sites have data from before and after harvesting (Bækken and Bratli, 1995); the results are presented in Table 3.3. The average phosphorus losses before harvesting were 7.8 g P/daa/year and increased to an average of 28.3 g P/daa/year after harvesting; an increase of 3.7 times. Also De Wit et al. (2014) found an increase of 3.8 (from 1.9 to 7.3 g P/daa/year) between before and after harvesting. The absolute P losses at Langtjern after harvesting were lower than the after harvesting sites studied by Bækken and Bratli (1995). Andebu II which had losses before harvesting of showed a P loss of 10 g P/daa/year after harvesting.

Major seasonal and interannual variations have been observed in phosphorus runoff (Vandsemb 2006; Bækken and Bratli 1995). The variation is largely controlled by the amount of runoff (Vandsemb 2006). Bækken and Bratli (1995) found that the observed variation appeared to be only partially geographically and rainfall dependent.

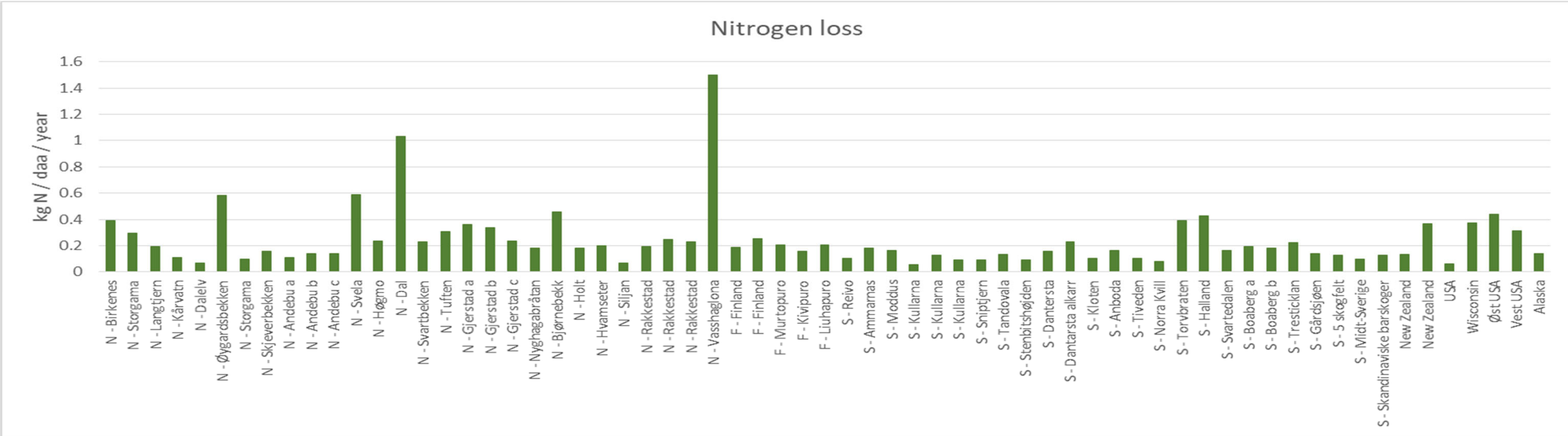

Figure 3.1: Nitrogen losses in Norwegian (N), Finnish (F), Swedish (S) and other forest areas (Appendix 1) (Garmo and Skancke, 2018; De Wit and Wright, 2008a; Vandsemb, 2006; Bækken and Bratli, 1995).

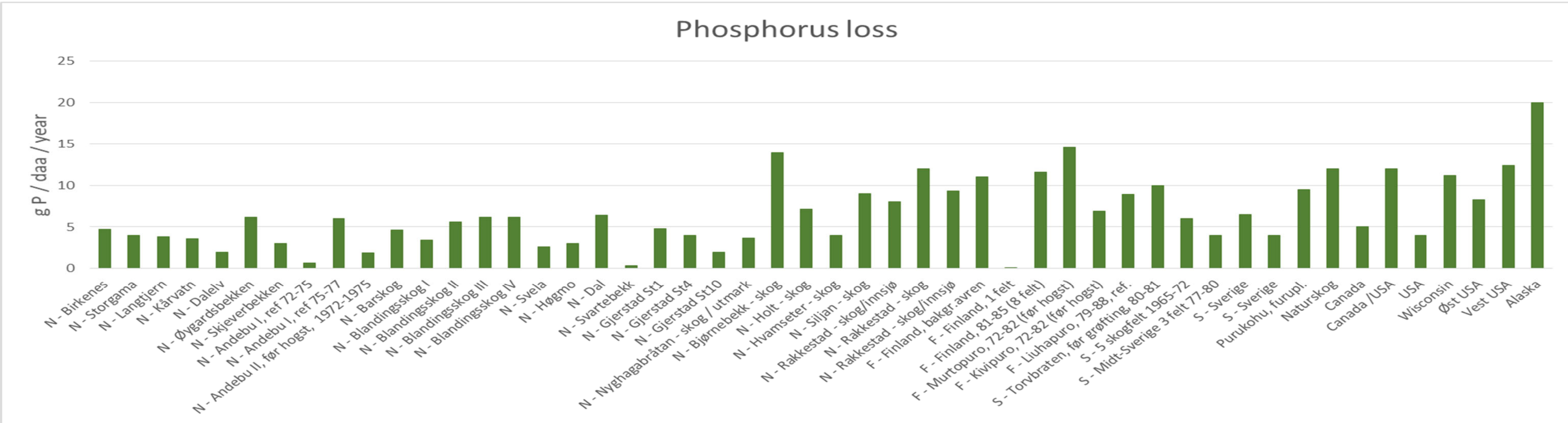

Figure 3.2: Phosphorus losses in Norwegian (N), Finnish (F), Swedish (S) and other forest areas (Appendix 1) (Garmo and Skancke, 2018; Vandsemb, 2006; Bækken and Bratli, 1995).

Table 3.3: Phosphorus losses in Nordic forested areas before and after harvesting (Bækken and Bratli, 1995).

	Period		P g/daa/year		
	Before harvest	After harvest	Before harvest	After harvest	
Norway					
Andebu II	1972-1975	1975-1977	1.9	10	Haveraaen, 1981
Finland					
Murtopuro	1972-1982	1982-1985	14.6	65.5	Ahtiainen, 1992
Kivipuro (with riparian zone)	1972-1982	1982-1985	6.9	9.3	Ahtiainen, 1992

3.3 Suspended sediments from forests

Loss of soil particles (suspended sediments, SS) from forested catchments in Norway and Finland ranged from 0.02 to 2.20 kg SS/daa/year, on average 0.76 kg SS/daa/year (Table 3.1 and Figure 3.3). In the Norwegian studies, the soil loss was between 0.25 - 2.20 kg SS/daa/year with an average of 0.95 kg SS/daa/year. For the Finnish catchments the average loss of suspended sediments was much lower than the Norwegian ones, with 0.28 kg SS/daa/year (range 0.02 – 0.4 kg SS/daa/year) (Figure 3.3).

The high losses of suspended sediments (2.2 kg/daa/year) from the catchment Bjørnebekk Skog may be related to tree harvest (clear-cutting) on 33 % of the area. Also at Nyhaga, tree harvesting was done during the monitoring period and losses of suspended sediments were relatively high (1.2 kg/daa/year). It should be noted that the measurements in Nyhaga catchment were based on continuous flow proportional samples, which have a better representation of high flow events compared to grab samples in the other studies.

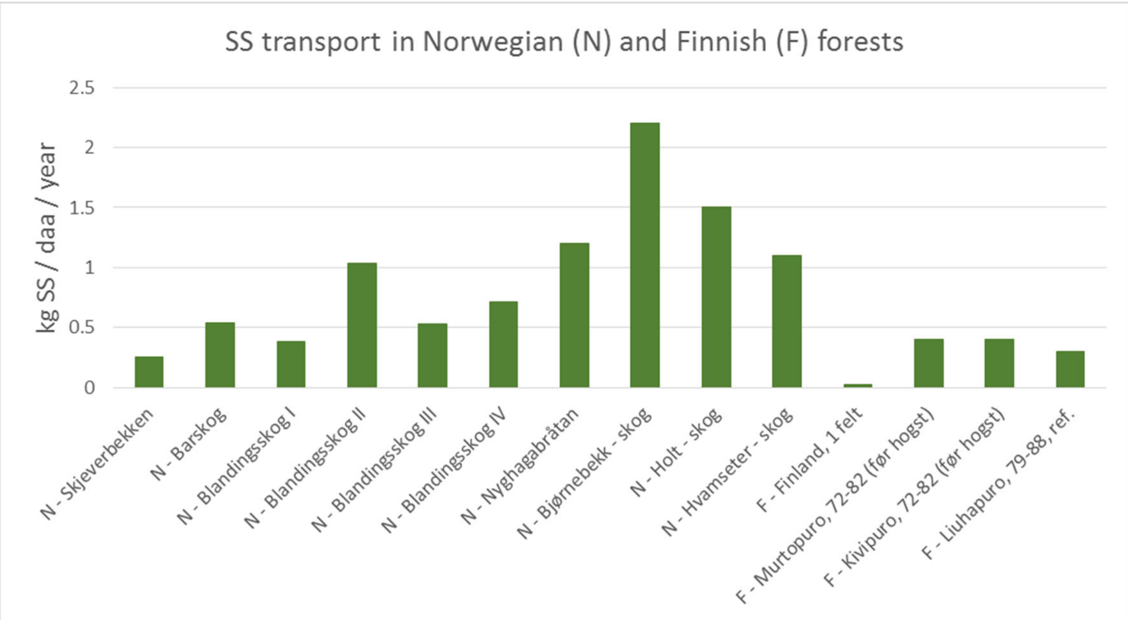

Figure 3.3: Suspended sediment losses in Norwegian and Finnish forest areas. Sites named N are Norwegian, sites named F are Finnish (Appendix 1) (Vandsemb, 2006; Bækken and Bratli, 1995).

Table 3.4: Sediment losses in Nordic forested areas before and after harvesting (Bækken and Bratli, 1995).

	Period		P g/daa/year		
	Before harvest	After harvest	Before harvest	After harvest	
Finland					
Murtopuro	1972-1982	1982-1985	0.4	0.6	Ahtiainen, 1992
Kivipuro (with riparian zone)	1972-1982	1982-1985	0.4	0.4	Ahtiainen, 1992

4 Nutrient and sediment losses from agriculture

The purpose of cultivation of new land is to increase food production. In Norway, the other Nordic countries, the Baltic countries and many others, the impacts of agricultural land use on water quality are monitored in a network of catchments representing important agricultural production systems (Figure 4.1; Table 4.1). The monitoring started in the 1990s and comprises data on discharge, suspended sediment and nitrogen and phosphorus concentrations.

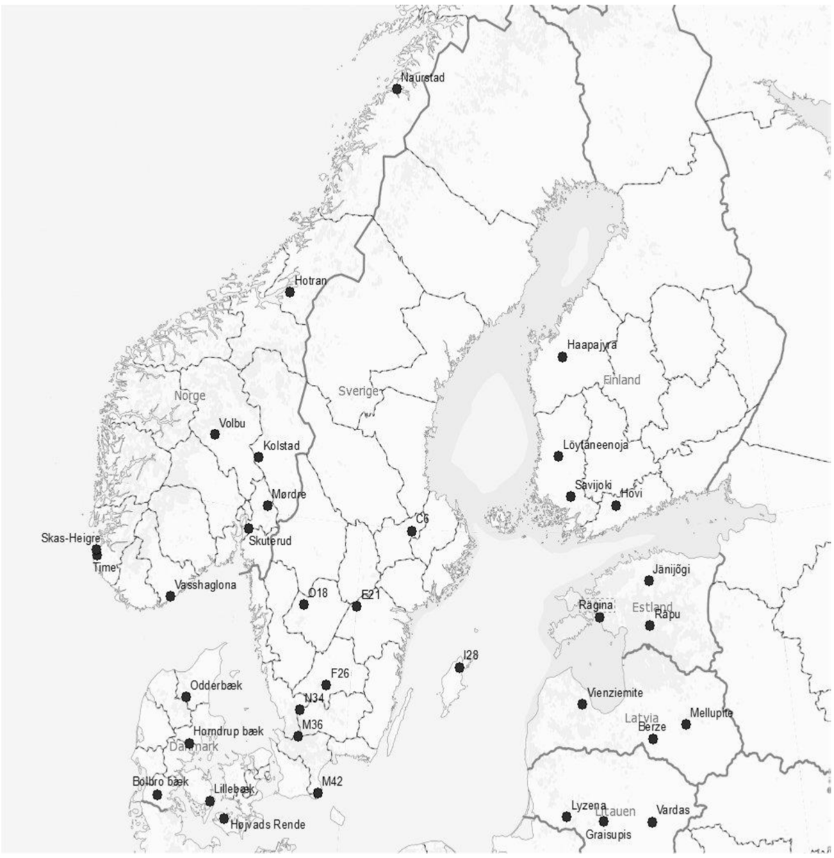

Figure 4.1: Monitoring network of 35 agricultural catchments in the Nordic and Baltic countries (Stålnacke et al. 2014).

Table 4.1: Characteristics of the 35 agricultural monitoring catchments (Kyllmar et al., 2014)

Country/ catchment	Area (km^{-2})	Agric. land [a] (%)	Dominant soil texture class on arable land according to USDA	Drained area (arable land) (%)	Production	Livestock density (AU ha^{-1})	Average temp. (°C)	Annual prec. (mm)
DENMARK (DK)								
Højvads Rende	9.8	65	Loamy sand	72	Cereals, sugarbeet	0.3	9.4	739
Lillebæk	4.7	89	Loamy sand	8	Cereals	1.0	9.3	833
Bolbro bæk	8.2	99	Sand	0	Cereals, grass, forage crops	1.3	8.9	1106
Horndrup bæk	5.5	82	Loamy sand	0	Cereals	1.2	8.5	919
Odderbæk	11.4	98	Sand	10	Cereals, grass, forage crops	1.4	8.4	939

NORWAY (NO)								
Skuterud	4.5	61	Clay loam, silt loam	100	Cereals	0.2	6.3	930
Mørdre	6.8	62	Silt, silt loam	100	Cereals	0.1	5.3	762
Kolstad	3.1	68	Loam, loamy sand	100	Cereals	1.3	4.4	751
Vasshaglona	0.7	65	Silt loam, loamy sand	100	Vegetables, potatoes	1.7	8.2	1429
Time	1.0	86	Loamy sand	100	Grass, pasture	2.3	8.5	1278
Skas-Heigre	28.3	84	Loamy sand	100	Grass, cereals	-	8.4	1237
Volbu	1.7	43	Loamy sand	100	Grass	0.4	2.9	587
Hotran	20.0	58	Silty clay loam, silt loam	100	Cereals, grass	-	6.1	997
Naurstad	1.5	35	Peat on loamy sand	100	Grass, pasture	0.6	5.2	1258
SWEDEN (SE)								
M42	8.2	92	Sandy loam, loam	100	Cereals	<0.1	7.7	709
M36	7.9	86	Clay, sandy loam	88	Cereals, grass, potatoes	0.3	7.6	719
N34	13.9	85	Sandy loam, silt loam	93	Cereals, grass, potatoes	0.4	7.2	886
F26	1.8	71	Sandy loam		Grass	0.9	6.2	1066
O18	7.7	92	Clay	100	Cereals	0.1	6.1	655
E21	16.3	89	Sandy loam	95	Cereals	0.2	6.0	506
I28	4.8	78	Sandy loam	99	Cereals, grass, potatoes	0.3	6.9	587
C6	33.1	59	Clay loam	95	Cereals	<0.1	5.5	623
FINLAND (FI)								
Haapajyrä	6.1	58	Clay, peat	-	Cereals	<0.1	4.5	545
Löytäneenoja	5.6	77	Clay, sand	-	Cereals, potatoes, sugar beet	<0.1	5.1	604
Savijoki	15.4	39	Clay, loamy sand	-	Cereals	<0.1	5.8	644
Hovi	0.1	100	Heavy clay	100	Cereals, oilseeds	0.0	4.7	679
ESTONIA (EST)								
Räpu	24.9	61	Sandy clay loam, loamy sand, peat	80	Cereals, oilseeds, grass	0.5	5.3	717
Rägina	21.1	53	Sandy clay loam, peat	100	Cereals, oilseeds, grass	0.2	5.6	662
Jänijõgi	18.4	59	Sandy clay loam, sandy loam	-	Grass, cereals	-	5.8	705
LATVIA (LV)								
Vienziemite	5.9	78	Sandy loam	100	Grass, pasture	0.5	5.7	715
Berze	3.7	98	Silty clay loam	100	Cereals, oilseeds	-	7.4	589
Mellupite	9.6	69	Loam	100	Cereals, grass	0.1	6.4	666

LITHUANIA (LT)								
Lyžena	1.7	97 [b]	Sandy loam	100	Pasture, cereals	0.6	6.5	706
Graisupis	14.2	69 [b]	Loam	100	Cereals, sugar beet	0.9	7.3	561
Vardas	7.5	73 [b]	Loamy sand	100	Pasture, cereals	0.4	7.2	661

[a] Agricultural land with harvested crops
[b] Animal unit per hectare
[c] Including permanent pasture

4.1 Nitrogen losses from agriculture

On average, for the agricultural catchments monitored in Norway by the JOVA programme, the fraction of applied nitrogen leached to water is calculated to 22 % based on long-term time series (Bechmann et al., 2012). The leaching of nitrogen is highest in sandy soils and on soils where subsurface drainage constitutes a large share of total runoff (Kværnø and Bechmann, 2010).

Stålnacke et al. (2014) summarised results on nitrogen losses in 35 catchments dominated by agriculture in the Nordic and Baltic countries. The text below is mainly based on their summary:

The investigated catchments showed large variation in nitrogen concentrations and losses (Figure 4.2), with a large interannual variability in all catchments. The overall range in total nitrogen losses was 0.7 – 10.2 kg N/daa/year. Generally, the largest losses were observed in the Norwegian catchments and the largest average total nitrogen loss (10.2 kg/daa/year) for one single catchment was observed in Vasshaglona in Agder County, which is characterised by intensive arable crops (e.g. vegetables) on sandy soils, high precipitation, high specific runoff (1200 mm), and for Norwegian conditions relatively high nutrient application rates, both manure and mineral fertiliser (Bechmann, 2014).

Lowest average total nitrogen losses, less than 1 kg/daa/year, were observed in three catchments in Denmark, Latvia and Lithuania. The two catchments in the Baltic states mainly represent pasture land with relatively low fertiliser application and low water discharge. The Danish catchment had very high retention in the aquifer, apparently related to nitrate reduction by pyrite, as high iron and sulphate concentrations are measured in the groundwater (Windolf et al. 2012).

In Norway, the catchments with extensive livestock (Naurstad and Volbu) are among the catchments with the lowest nitrogen losses (Table 4.2). These two catchments are representing low intensity agricultural production with low nitrogen application and low production. The within-country variations are largest for Norway and smallest for Finland. A probable reason for the high nitrogen losses and variability among the Norwegian catchments is the wide range of climatic and hydro-meteorological conditions in this country as illustrated by runoff ranging from 200 mm to more than 1200 mm (Stålnacke et al. 2014). Overall, around 54% of the total variability in the average total nitrogen losses in the 35 catchments can be explained by water discharge (Stålnacke et al. 2014).

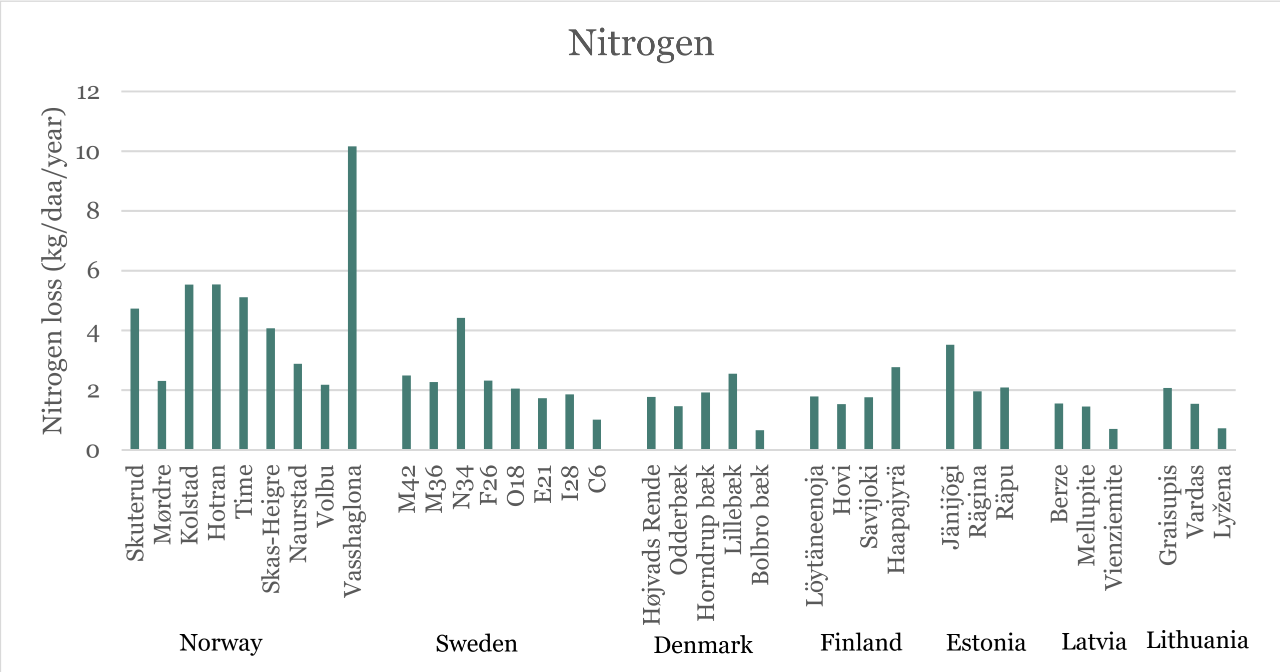

Figure 4.2: Average nitrogen losses in Nordic and Baltic agricultural catchments (adapted from Stålnacke et al. 2014).

4.2 Phosphorus losses from agriculture

Pengerud et al. (2015) summarised data from monitoring in 35 small agricultural catchments in the Nordic and Baltic countries, and the text in this section is mainly based on their work. The results showed substantial differences in the levels of phosphorus concentrations and losses among the catchments (Pengerud et al. 2015; Figure 4.3). Many of the Norwegian catchments have high runoff in combination with high concentrations of total phosphorus (Tot P) and dissolved reactive phosphorus in stream water, which results in large P losses.

The highest mean annual losses of total phosphorus from agricultural land was observed in the Norwegian catchment Vasshaglona (0.75 kg Tot P/daa/year), followed by Hotran (0.4 kg Tot P/daa/year) and Naurstad (0.37 Tot P/daa/year). The Vasshaglona catchment is characterised by intensive arable crops (e.g. vegetables) on sandy soils, high precipitation, high specific runoff (1200 mm), and for Norwegian conditions relatively high application of phosphorus in both manure and fertiliser (high rate over time), which results in a high soil phosphorus status (Bechmann, 2014). Due to comprehensive soil tillage, the erosion risk is high, which contributes to large losses of particulate phosphorus (Hauken & Kværnø, 2013). In Hotran, erosion also contributes significantly to total phosphorus losses (Bechmann, 2014). The Naurstad catchment is dominated by peat on loamy sand, with a low P adsorption capacity in the subsoil (Bechmann, 2014). The Naurstad catchment was also the catchment with the highest mean annual concentration of dissolved reactive phosphorus in stream water (161 µg P/L; Table 2). These high concentrations have been explained by a relatively high P surplus (P applied minus P removed in yield; on average 1.2 kg P/daa/year) in production, which may be lost through the soil due to its low sorption capacity (Bechmann, 2014).

High mean annual losses and concentrations of total phosphorus were also observed in O18 (Sweden), Hovi and Savijoki (both Finland). For O18, the high concentrations of total phosphorus in stream water has been explained by erosion losses of phosphorus due to a high clay content in the agricultural soils (approximately 60%) and a large proportion of annual crops (97%; Kyllmar et al. 2014), which generally have more soil management than perennial crops. The Hovi and Savijoki catchments are both located in southern Finland, a region that experienced mild winters towards the end of the study period, resulting in increased in-stream total phosphorus concentrations (Puustinen et al. 2007), due to higher erosion. Low contributions of dissolved reactive phosphorus to total phosphorus losses were observed in the Finnish catchment Haapajyrä (7%) and in the Norwegian catchment Mørdre (11%). This suggests that erosion may be a dominant transport pathway for phosphorus in these catchments, where particulate phosphorus contributes significantly to total phosphorus loss.

In Haapajyrä, the oxidation of sulphide-rich old marine deposits produces sulphuric acid in drainage water, resulting in low pH values and effective retention of phosphorus in aluminium and iron compounds (Vuorenmaa et al., 2002). The smallest total phosphorus losses were observed in E21 in Sweden and in Vienziemite in Latvia (both ~0.01 kg Tot P/daa/year). Both catchments showed low in-stream concentrations of total phosphorus, in combination with low water discharge. The soil in E21 is calcareous and thus calcium carbonates and iron oxides control phosphorus sorption (Castro & Torrent, 1998). The low concentrations of total phosphorus in Vienziemite can be explained by a low percentage (5–10%) of arable land within the catchment, where the agricultural land is mainly used as pasture or perennial grassland.

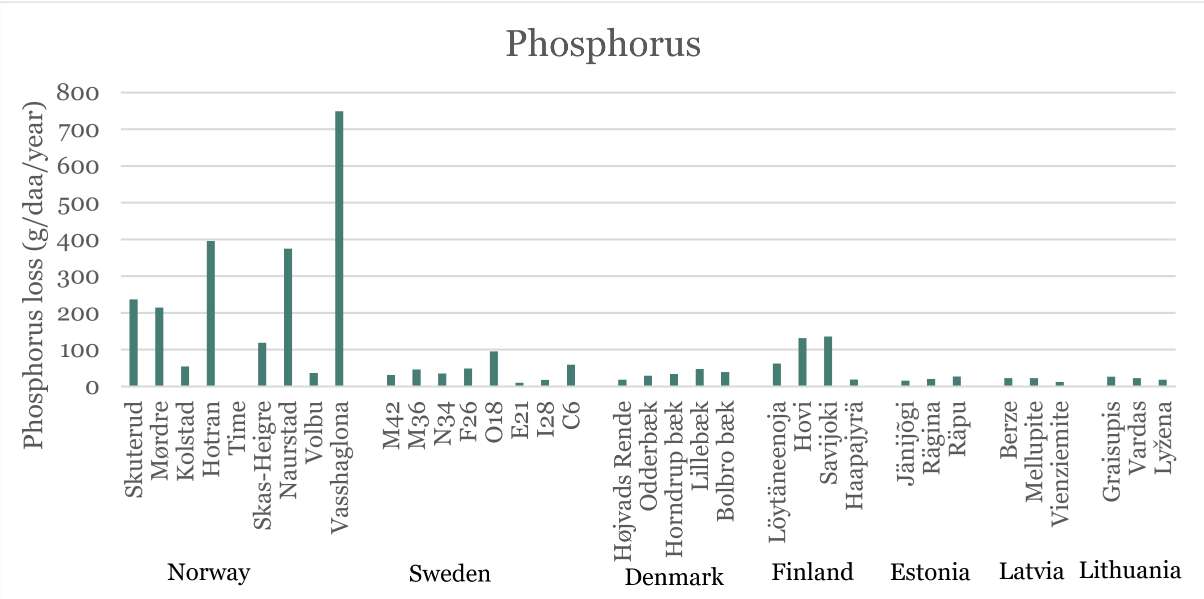

Figure 4.3: Average phosphorus losses in Nordic and Baltic agricultural catchments (Pengerud et al. 2015).

4.3 Suspended sediments from agriculture

Bechmann et al. (2017) showed that in the studied Norwegian agricultural catchments, the losses of suspended sediments from agricultural land varied between 9 and 318 kg/daa/year, with an average of 100 kg/daa/year (Table 4.2; Figure 4.4). There is a large difference between the lowest and highest loss of suspended sediments. The catchment with the highest suspended sediment loss is Hotran (318 kg/daa/year), followed by Mørdre (232 kg/daa/year), Vasshaglona (158 kg/daa/year) and then Skuterud (113 kg/daa/year). Hotran and Mørdre are located in areas with high erosion risks (Bechmann et al. 2017).

The average losses of suspended sediments are higher than 100 kg/daa/year in four of the Norwegian catchments and one of the Swedish catchments (Figure 4.4) out of the total 17 studied catchments. Loss of suspended sediments around <100 kg/daa/year may be seen as tolerable erosion (Verheijen et al., 2009). High losses of suspended sediments from agriculture occur from areas with arable crops with autumn ploughing, in rainy conditions, in sloping terrain and on easily erodible soil (Bechmann et al., 2017). Cereal production is dominating in Skuterud, Mørdre and Hotran with a varying share of the fields autumn ploughed each year. Intensive agriculture with production of potatoes and vegetables, like in Vasshaglona, can also cause great erosion. Reduced soil tillage leads to significantly reduced erosion (Bechmann et al., 2020). Production of grass is usually the mode of operation that gives the least loss of sediments, like in Time, Skas-Heigre, Naurstad and Volbu. Naurstad has silt below the peat soil and in the stream the soil is easily erodible and there may be high losses of suspended sediments and adverse conditions for fish downstream.

The Swedish catchment with the highest losses of suspended sediments (O18) is dominated by cereal production and is located in an area with clay soil and high erosion risk (Kyllmar et al., 2014).

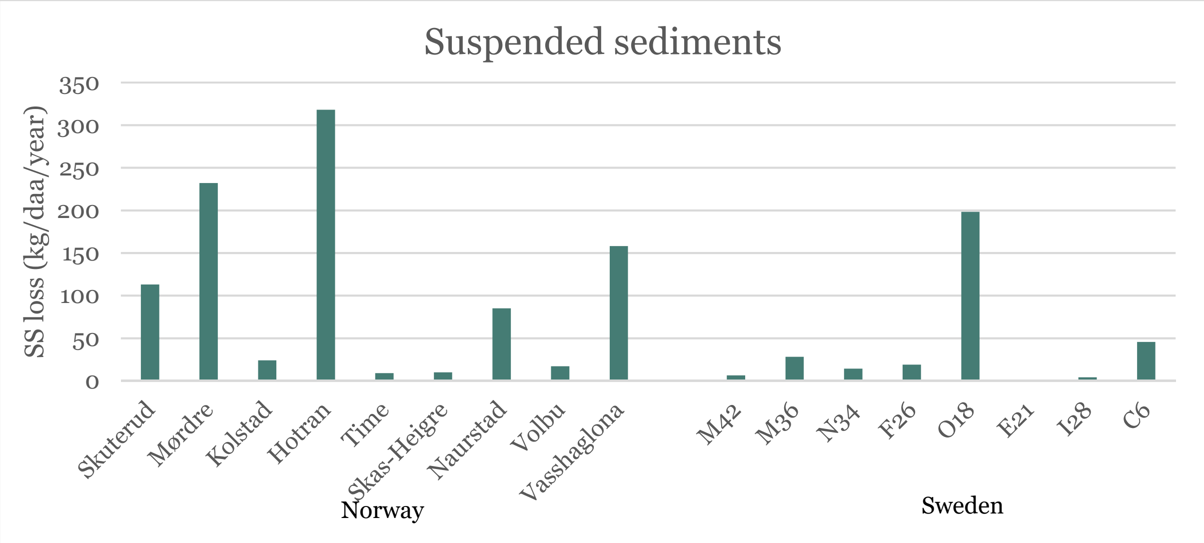

Figure 4.4: Average suspended sediment losses in Norwegian and Swedish catchments (Bechmann et al. 2017; Kyllmar, pers. comm.).

Table 4.2: Average suspended sediment (SS) losses and precipitation in the studied Norwegian catchments (Bechmann et al., 2017).

Catchment	Agriculture type	SS losses kg/daa/year	Precipitation (mm)
Skuterud	Cereals	113	898
Mørde	Cereals	232	729
Kolstad	Cereals and livestock	24	729
Hotran	Cereals and livestock	318	993
Naurstad	Extensive livestock	85	1254
Volbu	Extensive livestock	17	608
Skas-Heige	Intensive livestock	10	1259
Time	Intensive livestock	9	1345
Heia	Potatoes and vegetables	37	975
Vasshaglona	Potatoes and vegetables	158	1472

Like nitrogen and phosphorus losses there is a relationship within each catchment between runoff and suspended sediment loss (Figure 4.5). The strongest increase of suspended sediment loss related to increasing runoff is observed in Mørdre, followed by Naurstad and then Skuterud. In Volbu and Time, suspended sediment losses do not seem to be affected by higher runoff. Time even has a rather high annual precipitation, but since grassland is the dominating crop, most of the soil is covered and protected from erosion all year round.

Erosion is highest on soils with steep and long slopes. Soil that requires levelling is particularly exposed to erosion. Surface runoff occurs especially on low permeable soils (Grønlund et al., 2013). In Naurstad (extensive livestock), silt is covered with marsh and is exposed to erosion in the stream. In Hotran (cereals and livestock), some erosion has also been recorded in the stream bank, which may be the cause of the high loss of suspended sediments. In Vasshaglona, the concentration of suspended sediments is not particularly high, but large runoff, light sandy soil and low vegetation cover for much of the year due to vegetable production result in large losses of soil from agricultural areas.

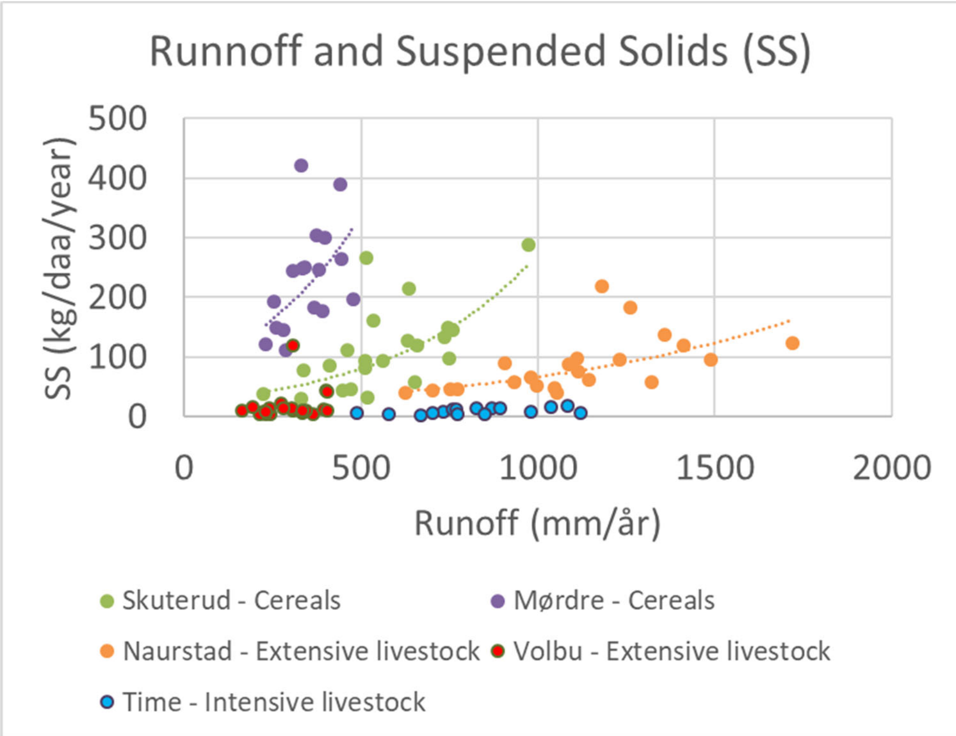

Figure 4.5: Relation between annual runoff and annual suspended sediment loss for a selection of studied agricultural catchments in Bechmann et al. (2017).

5 Nutrient and sediment losses in the transition phase

The transition phase is defined as the period when the forested areas or outfields are transformed to agricultural land.

Clearcutting using stem-only harvesting (SOH) is the most common forest harvesting practice in Norway. Land cleared for new cultivation, however, is considerably more altered than after SOH. Roots are pulled up, and the soil is ploughed, and often drained. Hence, to assess the effects of new cultivation on water quality, it is also necessary to include the nutrient losses during the period when the area is cleared for cultivation.

In Finland and Sweden, stump harvesting following clearcutting has become more common in recent years (Björheden, 2006, Juntunen & Herrala-Ylinen 2011), because tree stumps are a potential bioenergy resource to replace fossil fuels and meet the targets for reduced CO_2 emissions. The forest industry in Finland initiated stump lifting for bioenergy purposes in the early 2000s, while the Swedish interest in stump harvesting started in 2005. However, stump harvesting is not practiced in Norway at present.

Research on the effects of stump harvesting on the environment indicates what environmental impact might be expected during the transition phase (phase 2), after removal of the forest. Short term (3-4 years) and long term (20-30 years) effects have been studied. This has mainly been done by taking soil and soil solution samples, therefore the results in this chapter discuss soil water quality observations, which cannot be directly translated to water quality changes in streams and rivers, but do indicate possible effects on nutrient leaching.

In the process of cultivation, harvesting of trees and removal of the whole tree including stumps will be carried out. Harvesting and removal of all tree stems in a stand often results in change in the soil organic carbon and nitrogen balance caused by a suddden input of harvesting residues (branches, twigs, and green needles), dying roots and attached mycorrhizal fungi. Furthermore, increased decomposition due to higher temperatures in the forest floor and possibly also mixing of organic material into the mineral soil by harvest machines has an impact on soil nitrogen and carbon balances.

Several factors, like the concentrations in soil solution, the amounts of harvest residues, and topographic and climatic conditions, affect the concentrations of suspended sediment and nutrients in runoff.

<u>Whole-tree harvesting: Piles and removal areas</u>

In whole-tree harvesting (WTH), crowns and branches are harvested for use for bioenergy. A common practice is to pile the crowns and branches in the forest for a period to allow the nutrient-rich needles to fall off and thus reduce nutrient depletion. Piles are limited to relatively restricted parts of harvesting sites, whereas the areas from which the crowns and tops are removed cover larger areas. Thus, concentrations in runoff from a clear-cut after WTH will reflect largely the concentrations in soil solution under WTH removal areas (WTH-removal) and only to a lesser degree the concentrations under WTH piles (WTH-pile). In practice, not all harvest residues are removed: the amount is commonly 60–80% under Nordic conditions (Helmisaari et al. 2011, Thiffault et al. 2015), although skilled machine operators might be able to remove more. Where residues have been removed directly after harvesting in order to form piles (the WTH-removal treatment), a lower nutrient input is likely, resulting in lower concentrations. However, this is not always observed (Clarke et al. 2018), as factors like topography may also affect soil solution concentrations.

The elevated nutrient leakage after harvesting often appears to last from 4 to 12 years depending on soil conditions (Wiklander, 1983; Rosen et al., 1989; in Bækken and Bratli, 1995). Also the results of Zetterberg et al. (2013) indicate that the effect of WTH on soil and soil solution concentrations is often temporary and site specific.

5.1 Nutrient losses during the transition

Data on the effect of the transition phase on nitrogen and phosphorus loss from soil to surface water could not be found directly. However, data on soil solution were used to illustrate the difference in nutrient concentrations under standing forest and forest after harvest.

5.1.1 Short term effects of forest harvesting on soil water quality

Soil solution

According to the mobile anion theory (Reuss and Johnson, 1986), the leaching of base cations through the soil is dependent on the concentration of anions. This concept, based on electroneutrality and the mobility of different anions in soil solution, dictates that changes in base cation leaching are only possible if there is a corresponding change in the anion flux. This theory is important for the following discussion of leaching in forest soils.

Soil solution chemistry in forest soils has been shown to be affected by harvesting, with increased leaching of nutrients such as nitrogen and base cations as important effects. Staaf & Olsson (1994), Hu (2000), Piirainen et al. (2004) and Clarke et al. (2018) all found similar results: Hu (2000) found higher nitrate (NO_3^-) and potassium (K) concentrations in soil water from mineral soils two to three years after harvesting, and Piirainen et al. (2004) observed that the phosphorus flux under the organic layer increased three times and the base cation flux two times after harvesting, while the flux of sulphate decreased.

Clarke et al. (2018) studied short term effects of WTH-pile and WTH-removal as well as SOH at two experimental Norwegian sites. They analysed soil water samples during a period of three years after harvesting. In the years after harvesting, Clarke et al. (2018) found increased soil solution concentrations of NO_3^-, total N, Ca, Mg and K at 30 cm depth, at Gaupen, a relatively dry site with gentle slopes in eastern Norway after SOH and WTH-pile, but less so after WTH-removal. At Vindberg, a wetter and steeper site in western Norway, peaks were often observed also at WTH-removal plots, which might reflect within-site differences in water pathways due largely to site topography.

The increased nutrient concentrations observed in the three- to four-year period after harvest are probably due to decomposition, leaching and mineralisation of harvesting residues, in particular needles, fine twigs, dead fine roots and soil organic matter under the residues, combined with a lack of uptake by trees after harvesting and removal, and a limited uptake by ground vegetation.

In WTH, harvesting residues may be left in piles for periods of some months, allowing nutrient-rich needles to fall off and thus increasing the return of nutrients to the ecosystem. Nutrient fluxes may therefore be greater under piles compared to soils without piles. Effects of removal of harvesting residues on soil solution chemistry may depend on how long the residues are left on-site after harvesting. Wall (2008) compared the effects of immediate removal of branches and removal after two months. Although piles were only a minor source of inorganic nitrogen, they were a significant source of organic nitrogen, as well as phosphorus, Ca, Mg and K. In the treatment with branch removal after two months but foliage left on-site, concentrations of nutrients at the base of the upper horizon were slightly higher than for the SOH treatment (Wall 2008). Clarke et al. (2018) found that there were tendencies to higher peak concentrations for NO_3–N, total nitrogen, Ca, Mg and K in WTH-pile relative to WTH-removal and also SOH at Gaupen, suggesting that nutrient concentrations will be greater where there have been residue piles. At Vindberg, on the other hand, this tendency was not always observed, although for $NH_4{}^+$, total nitrogen, K and DOC, the concentrations post-harvest were considerably higher at the WTH-pile relative to the WTH-removal and SOH treatments. However, the response to the WTH-removal treatment at Vindberg was similar to or higher relative to the SOH plots. This lack of difference may be related to the steep terrain and the oceanic climate in combination with relatively shallow soil and a buildup of large piles, and also to the disturbance to the soil being more severe at Vindberg because of practical difficulties in harvesting the piles on the steep terrain. This resulted in severe damage to vegetation on several WTH-pile plots at Vindberg (Økland et al., 2016, Clarke et al., 2018) followed by risk of severe erosion on the steep slopes.

Clarke et al. (2018) found reduced sulphate concentrations after all treatments, which might reflect the reduction in dry deposition that would follow harvesting due to the removal of the tree crowns, and which might not be compensated by increased release of sulphate from the residues (Piirainen et al. 2004). In addition, it is possible that a large-scale long-term trend of declining $SO_4{}^{2-}$ concentrations, previously observed in deposition and soil solution in Norway (Kvaalen et al., 2002) as well as in surface waters (Garmo et al., 2014), might have contributed to reduction trends.

The pH in soil solution generally decreases after harvesting (Staaf and Olsson, 1994; Clarke et al., 2018), possibly related to increased nitrification and nitrate leaching. However, this decrease is likely to be only temporary, as Staaf and Olsson (1994) found that pH in soil solution tended to recover after four to five years, and there were indications of an increase in pH in the final year at Gaupen for all treatments and at Vindberg for the SOH treatment (Clarke et al., 2018).

Short-term increases in soil solution nitrate concentration are often observed after forest harvest, even in N-limited systems. Futter et al. (2010) modelled nitrate leaching below the rooting zone as a function of site productivity. Using national forest inventories and published estimates of N attenuation in rivers and the riparian zone, they estimated effects of stem-only harvesting on nitrate leaching to groundwater, surface waters and the marine environment. Stem-only harvesting is a minor contributor to nitrate pollution of Swedish waters. Effects in surface waters are rapidly diluted downstream, but can be locally important for shallow well-waters as well as for the total amount of N reaching the sea. Harvesting adds approximately 8 Gg NO_3-N to soil waters in Sweden, with local concentrations up to 7 mg NO3-N l^{-1}. Of that, 3.3 Gg reaches the marine environment. This is 3% of the overall Swedish N load to the Baltic.

Generally, concentrations of nutrients increased for some years after harvesting, at least where there were harvesting residues.

<u>Topograpic and climatic conditions</u>
Both local topographic and climatic conditions may affect vegetation as well as soil processes after harvesting.

Many forests have the capacity for retaining water. In the study by Clarke et al. (2018), soil disturbance removed many bryophytes, including *Sphagnum* spp., which have a high capacity for retaining water. Water from areas with mixtures of piles, spread harvesting residues and open soil may therefore have been leached downslope to a larger extent at Vindberg than at Gaupen, and this may have affected nutrient concentrations in downslope WTH-removal areas. Thus, especially in an area with steep terrain, water pathways may be important for effects of harvesting on soil solution chemistry. These observations of between-site differences were attributed at least partly to differences in topography as well as climatic conditions.

Increased summer soil solution concentrations of nitrate, total nitrogen, Ca, Mg and K at 30 cm depth were found at Gaupen, the drier, less steep site in eastern Norway, in a three- to four-year period after SOH and WTH where there had been piles but not where residues had been removed after harvesting to make piles (Clarke et al. 2018). At Vindberg, the wetter, steeper site in western Norway, less clear differences in soil solution concentrations between treatments might reflect a more complex hydrology due to the topographic differences between the sites. There were between-site differences in the time trends for changes in soil solution chemistry, with effects generally appearing slightly later at Vindberg, possibly because of less favourable conditions for decomposition.

5.1.2 Long term effects of harvesting on soil water quality

Soil characteristics and seasonal effects
In a study of Zetterberg et al. (2013) in Sweden, the soil solution composition varied substantially between three study sites (Lövliden, Kosta and Tönnersjöheden) both during the first (2003-2005) and second (2008-2010) measurement period. One of the study sites, the Lövliden site, was relatively well-buffered compared with the two other sites of this study (Kosta and Tönnersjöheden), which could be seen in the soil characteristics. Deposition at Lövliden is less acidic than at the other sites, which probably accounts for Lövliden being less acidic and having a better buffering capacity against further acidification. Lövliden had higher pH and base saturation, and lower exchangeable Al. Seasonal differences were also present for most variables during the two time periods. Hence, the soil water concentrations depend both on site and season, creating a large variation in the dataset.

The concentration differences between study sites were statistically significant for all the measured ions except Mg^{2+} and K^{+}, which are both macronutrients and to a large extent internally circulated within the forest ecosystem. The average concentrations of Mg^{2+} and K^{+} varied in narrow ranges, independent of site and treatment. At Lövliden, the sum of organic anions ($RCOO^{-}$) and HCO_3^{-} constituted approximately 65% of the anions independent of treatment. Both these anions originate from biological activities. Based on the cations and anions in soil solution, it seems reasonable to assume that biological processes primarily govern the ion composition in soil solution at Lövliden.

At all sites, the concentrations of nitrate in the soil solution were very low, usually below the detection limit (Zetterberg et al. 2013). Hence, at these harvested forest sites nitrate is not an important anion in terms of cation leaching. However, nitrate may have been important for the base cation leaching during the clear-cut phase, a period not covered by Zetterberg et al. (2013). Although similar processes may have taken place at the three sites following harvesting, the degree of impact of WTH and SOH on the anion-associated cation leaching to the B/C-horizon is likely to be a function of the differences in deposition of anions between sites.

The export of nutrients and buffer capacity is higher during WTH compared with SOH, which affects the acid neutralising capacity. Increased losses of soil base cations following WTH have been documented in many studies (Zetterberg et al. 2013). Even 27–30 years after harvest, soil solution concentrations of calcium were 40% lower in WTH plots compared with SOH plots. The removal of logging residues with WTH may lead to soil and surface water acidification by lowering the amount of buffering base cations (Zetterberg et al. 2013). The results of Zetterberg et al. (2013) indicate that the effect of WTH on soil and soil solution concentrations is site specific. More surprisingly, the greatest effects on soil and soil solution were observed at the well-buffered site where the loss of calcium during WTH is less likely to lead to acidification effects. The WTH and SOH treatment effects on soil solution at the more acidic sites in southern Sweden were much smaller and probably not large enough to fully counterbalance the general recovery from acidification during the study period.

The main treatment differences had largely disappeared 32–35 years after harvest although site specific treatment differences were still measurable at the well-buffered site in northern Sweden.

Acidification and effect on water quality
Data for the long-term acidification effects of WTH on soil water, groundwater and surface water are scarce. It is clear that acidification effects of forest harvesting (SOH or WTH) on aqueous media largely depend on the leaching of mobile anions, primarily nitrate produced during nitrification (e.g.

Dahlgren and Driscoll, 1994; Hendrickson et al., 1989; Neal et al., 1992). However, the long-term decreasing trends in Ca^{2+} concentrations in soil solution and surface waters in forested areas of Sweden (Löfgren et al., 2009a, 2011; Löfgren and Zetterberg, 2011) and elsewhere in the northern hemisphere (Skjelkvåle et al., 2005; Stoddard et al., 1999) have generally been attributed to reduced SO_4^{2-} deposition and lower soil solution concentrations of the counter-balancing SO_4^{2-} ion. Similar trends have been observed in poorly buffered Swedish streams subjected to low doses of forest soil liming (3 ton ha^{-1} or 60 kmolc⁻ Ca^{2+} ha^{-1}), indicating low or negligible coupling between the exchangeable Ca^{2+} levels in soils and the acidity trends of streams in a decadal perspective (Löfgren et al., 2009a). Hence, WTH may acidify the soils by reducing the exchangeable Ca^{2+} pools, but the impact on the Ca^{2+} concentrations in aqueous media may be less severe in cases where WTH does not lead to an increase in counter-balancing anions.

Stump harvesting

A study by Persson et al. (2017) presented soil sampling results 25 years after harvest. The main outcome showed that stump harvesting resulted in a non-significant trend of lower carbon (6.6% lower) and nitrogen (3.5% lower) pools in the humus layer (to a soil depth of 20 cm) than after patch scarification with the stumps retained. The C/N ratio was not significantly affected by the two treatments (Persson et al., 2017). No significant differences between treatments could be detected for the soil organic nitrogen pools, which largely followed the soil carbon pools. There was no significant difference between treatments in inorganic nitrogen pools.

5.2 Sediment losses during transition

Under various weather conditions, heavy machinery used for forest harvesting of any type may damage the soil and create compaction and the risk for erosion in the wheel tracks. Furthermore, cultivation of new land implies disturbances of the soil that may increase the soil loss and give increased siltation in rivers and lakes.

For example in Segeråga in northern Norway, comprehensive sedimentation of mineral particles and consequently reduced quality of the stream with regards to fish were registered after cultivation that had lasted since the 1970s (Bergan and Aanes, 2017). The effects of cultivation may not only be temporary during the transition, but may continue after cultivation depending on the measures in place. Erosion in stream banks also seemed to have increased in Segeråga due to cultivation. Measures should therefore include both erosion prevention on fields and in drainage channels, and to prevent access to the stream by grazing catle. Sedimentation ponds and vegetation along open channels and streams were also recommended to trap sediments (Bergan and Aanes, 2017).

Another example is the changes in the catchment of the Semselva in Mid-Norway, in the form of lowering of two lakes, Lundavatnet and Lømsen, ditching, cultivation, mass withdrawal, soil erosion and increased nutrient supply. These changes altered the water quality and habitat quality so much that the population of river mussels is presently in danger of disappearing (Larsen, 2019).

In a study comparing WTH and SOH at the two Norwegian sites mentioned above, Gaupen and Vindberg (Økland et al., 2016; Clarke et al., 2018), severe soil disturbance occurred. Even though direct traffic of heavy machinery was avoided inside the study plots, the harvesting of the WTH residues using the crane boom caused some soil disturbance that could have contributed to increased decomposition. Soil disturbance was especially severe where piles had been collected at Vindberg, the site with steepest terrain and high precipitation.

The loss of suspended sediments during and after harvest is not a linear process. Bormann et al. (1974) showed that the loss of suspended sediments after harvest reaches its maximum after a couple of years due to increased erodibility in the catchment. The conditions during harvest are important for this process and the local climate and geology may influence this pattern, which may deviate significantly from the described. Harvest machinery may cause huge erosion if no precautions are taken.

Mechanical site preparation (in Norwegian, "markberedning") is sometimes used after harvesting and before planting to improve the growth conditions for the new trees. There are several different methods (Hanssen, 2017), and in general methods that disturb the soil least are likely to lead to smaller losses of soil.

The different methods of tree harvesting may be used both as part of normal forestry practice and when forest is harvested for cultivation, and gives as such no additional effect of cultivation in the transition phase. However, in the case of cultivation, large changes will occur over all of the cultivated area and the changed conditions will of course continue.

Drainage with open ditches may also cause increased sediment transport, especially with cultivation on silty, highly erodible soils; eventually underlaying peatlands. New ditching of wetlands is no longer allowed in Norwegian forestry, although maintenance of existing ditches can still be done.

5.3 Losses of organic carbon during transition

Locally, there is an increase in loss of organic carbon after tree harvest due to decomposition of organic matter from tree residues left on and in soil, and of soil organic matter (Neal et al. 1992; Steedman, 2000). This change will affect the water colour and temperature. The effect of tree harvest on organic matter concentrations disappears at the large catchment scale because of the dominating impact of climate (Lepistö et al. 2014). The immediate effect of drought is that organic matter will be mobilised (Lepistö et al. 2014).

An additional effect of organic matter is the binding of heavy metals in organic complexes (Lasota et al. 2020). They may be mobilised together with organic matter and leach into the recipient water body. However, heavy metal concentrations are normally low in Norwegian forests, including after harvest, except near some local pollution sources.

6 Effects of cultivation of new land on water quality

The total effect of cultivation of new land on water quality includes the difference in sediment and nutrient losses between agricultural land use and forest/outfields (long term effects) as well as losses occurring during the transition phase (short to long term effects). As noted in the methodology chapter, the differences between losses from agricultural and forested catchments are largely due to the different soil, geological and climaticconditions.

6.1 Nutrient and suspended sediment losses from forests versus agriculture

There is a general difference in losses of suspended sediment and nutrients from forest and outfields and agricultural land (Table 6.1). In the presented Nordic studies, the losses of nitrogen from agricultural areas were on average approximately 12 times that of forested areas, and the losses of phosphorus were approximately 19 times higher for agriculture than for forest. The largest difference between these two land use types was for suspended sediments, where the losses were on average 96 times higher for agriculture than for forest (Table 6.1).

Table 6.1: Average nitrogen, phosphorus and suspended sediment (SS) losses in forest and agriculture catchments. (Garmo and Skancke, 2018; De Wit and Wright, 2008a; Vandsemb, 2006; Bækken and Bratli, 1995; Stålnacke et al. 2014; Pengerud et al. 2015; Bechmann et al. 2017)

	Nitrogen average (range) kg Tot N / daa /year	Phosphorus average (range) g Tot P / daa / year	Suspended sediments average (range) kg SS / daa / year
Forestry			
Norwegian forest	0.3 (0.07-1.50)	5.07 (0.31-13.90)	0.95 (0.25-2.20)
Nordic forest	0.24 (0.06-1.50)	5.75 (0.02-14.60)	0.76 (0.02-2.20)
Field studies – before harvesting	0.13 (0.09-0.20)	7.80 (1.90-14.60)	0.40 (0.40-0.40)
Field studies – after harvesting	0.49 (0.23-0.95) Increase of 3.7 times	28.27 (9.30-65.50) Increase of 3.6 times	0.50 (0.40-0.60) Increase of 1.25 times
Agriculture			
Norwegian agriculture*	5.2 (2.1 – 9.8)	285 (41 – 750)	100 (9 – 318)
Nordic agriculture*	2.8 (0.66 – 9.8)	108 (9 –750)	73 (1.5 – 318)

*The data of the Norwegian agriculture catchments in this section are collected from Bechmann et al. 2017.

There are large differences in losses of sediments and nutrients between different agricultural production systems (chapter 4). Hence, the effect of changing land use from forest/outfields to agriculture depends on which agricultural production system is planned, and also on the climate, topography and soil type at the location to be cultivated.

High precipitation and runoff contribute to the relatively high losses of suspended sediments, phosphorus and nitrogen in Norway compared to other Nordic countries (Table 6.1). For suspended sediment and phosphorus the hilly landscape also contribute to higher losses. Due to the high losses in Norway, we have focused on results from studies in Norway in the following evaluation of effects on water quality of cultivation of new land.

6.1.1 Nitrogen

Table 6.2: Average total nitrogen loss in the studied Norwegian catchments with forest and agriculture.

Catchment	Type of forest	kg N/daa/year
Storgama	Mountain area, peat, 11% low prod. forest	0.10
Storgama	Mountain area, peat, 11% low prod. forest	0.30
Birkenes	Forest, high production	0.39
Langtjern	Mountain area, peat, 5% low prod. forest	0.20
Kårvatn	Mountain area, peat, 18% low prod. forest	0.11
Dalelv	Mountain area, peat, 20% low prod. forest	0.07
Øygardsbekken	Mountain area, peat, 4% low prod. forest	0.58
Svartebekken	Forest and outfields	0.23
Tuften	Forest and outfields	0.31
Holt	Forest, high production, no harvest	0.18
Hvamseter	Forest, high-medium prod.	0.20
Siljan	Forest, medium prod.	0.07
Rakkestad	78% forest, 89% low produc.	0.19
Rakkestad	78% forest, 89% low produc.	0.23
Rakkestad	86% forest, 70% low produc.	0.25
Nyghagabråtan	Forest, some harvest and peat and old grassland	0.18
Bjørnebekk	Forest, high prod., ca. 30% harvest (clear-cut)	0.46
Dal	Prod. forest with clear-cut areas	1.03
Catchment	**Agriculture type**	**Kg N/daa/year**
Skuterud	Cereals	4.8
Mørde	Cereals	2.4
Kolstad	Cereals and livestock	5.5
Heia	Cereals, vegetable and potato	9.5
Vasshaglona	Cereals, vegetable and potato	9.8
Hotran	Cereals and livestock	5.2
Naurstad	Extensive livestock	2.8
Volbu	Extensive livestock	2.1
Skas-Heige	Intensive livestock and grass	3.9
Time	Intensive livestock and grass	5.6

The results for the Norwegian data show a large difference in nitrogen losses between sites with agricultural land use and sites with forest, outfields and peat. Nitrogen was on average 0.3 kg N/daa/year in forest and on average 5.2 kg N/daa/year in agriculture, which means that losses from agriculture were about 17 times higher than the losses from forest (Table 6.1). Even in the highly productive forest area with tree harvest in Dal, losses of nitrogen (1.03 kg/daa/year) were still only half of the nitrogen losses from agricultural land use with extensive livestock production and grassland (2.1 kg/daa/year; Table 6.2). The lowest observed nitrogen losses in agriculture (2.1 kg N/daa/year in the study of Bechmann et al. 2017; Table 6.2) were still seven times higher than the average loss in forest.

6.1.2 Phosphorus

Table 6.3: Average total phosphorus losses in the studied Norwegian catchments with forest and agriculture.

Catchment	Type of forest	g P/daa/year
Storgama	Mountain area, peat, 11% low prod. forest	4.0
Birkenes	Forest, high production	4.7
Langtjern	Mountain area, peat, 5% low prod. forest	3.8
Kårvatn	Mountain area, peat, 18% low prod. forest	3.6
Dalelv	Mountain area, peat, 20% low prod. Forest	2.0
Øygardsbekken	Mountain area, peat, 4% low prod. Forest	6.2
Svartebekk	Forest and outfields	0.3
Holt	Forest, high production, no harvest	7.1
Hvamseter	Forest, high-medium prod.	4.0
Siljan	Forest, medium prod.	9.0
Rakkestad	78% forest, 89% low produc.	8.0
Rakkestad	78% forest, 89% low produc.	9.3
Rakkestad	86% forest, 70% low produc.	12.0
Nyghagabråtan	Forest, some harvest and peat and old grassland	3.7
Bjørnebekk	Forest, high prod., ca 30% harvest (clear-cut)	13.9
Dal	Prod. forest with clear-cut areas	6.5
Catchment	**Agriculture type**	**g P/daa/year**
Skuterud	Cereals	252
Mørde	Cereals	340
Kolstad	Cereals and livestock	66
Heia	Cereals, vegetable and potato	361
Vasshaglona	Cereals, vegetable and potato	750
Hotran	Cereals and livestock	417
Naurstad	Extensive livestock	365
Volbu	Extensive livestock	41
Skas-Heige	Intensive livestock and grass	111
Time	Intensive livestock and grass	144

With respect to phosphorus there is an increase from on average 5.07 g P/daa/year in forestry to on average 285 g P/daa/year in agriculture, this is more than 56 times the loss from forestry (Table 6.1). The maximum phosphorus loss from forest observed by Vandsemb (2006) (13.9 g P/daa/year, Table 6.3) is nearly three times lower than the minimum measured phosphorus loss in agriculture (Bechmann et al., 2017) (41 g P/daa/year, Table 6.3).

6.1.3 Suspended sediments

Losses of suspended sediments (SS) from forest and outfields are very low compared to losses measured from agricultural areas (Table 6.4). In the Norwegian studies, the soil losses are between 0.25 - 2.20 kg SS/daa/year in forest areas, which are very low compared to losses from agricultural areas (variation between about 9-318 kg SS/daa/year) (Table 6.1 & 6.4). On average this results in an increase of more than 106 times. From the average losses of suspended sediments in Norwegian forest (0.95 kg SS/daa/year) there is at least an increase in losses of suspended sediments of nine times compared with Time (9 kg SS/daa/year), the catchment with the lowest losses of suspended sediments.

Table 6.4: Average suspended sediment (SS) losses in the studied Norwegian catchments in forest and agriculture.

Catchment	Type of forest	Kg SS/daa/year
Skjeverbekken		0.25
	Coniferous forest	0.54
	Mixed forest I	0.38
	Mixed forest II	1.04
	Mixed forest III	0.53
	Mixed forest IV	0.71
Nyghagabråtan	Forest, some harvest and peat and old grassland	1.2
Bjørnebekk	Forest, high prod., ca 30% harvest (clear-cut)	2.2
Holt	Forest, high production, no harvest	1.5
Hvamseter	Forest, high-medium prod.	1.1
Catchment	**Agriculture type**	**kg SS/daa/year**
Skuterud	Cereals	113
Mørde	Cereals	232
Kolstad	Cereals and livestock	24
Heia	Cereals, vegetable and potato	37
Vasshaglona	Cereals, vegetable and potato	158
Hotran	Cereals and livestock	318
Naurstad	Extensive livestock	85
Volbu	Extensive livestock	17
Skas-Heige	Intensive livestock and grass	10
Time	Intensive livestock and grass	9

6.2 Transition from forest to agriculture

In addition to long term effects of transforming forests/outfields into agriculture, there is also an effect in the transition phase when the new land is cultivated. This comprises harvesting of stems and stumps, and removal of all residues. The data on harvesting forest showed that nitrogen and phosphorus losses increased by approximately 4 times, and suspended sediments by approximately 1.25 times (although from limited data) during tree harvesting compared to standing forest. As opposed to new cultivation, forest is only harvested a few times during a century, which means that on average only small parts of the forest are harvested each year.

The increase in nitrogen losses seems to depend on the amount of trees and branches left on-site. Harvesting of forest (stem-only) can increase the total nitrogen losses maximum 6.8 times as compared to nitrogen losses in undisturbed forest. During the transition for cultivation of land not only the stems, but also the stumps will be harvested to make the site suitable for agricultural land use. Thereby, the biological material left on-site will be low and mineralisation and release of nutrients correspondingly low. With respect to phosphorus losses the maximum observed increase due to tree harvesting was 10 g P/daa/year, an increase of 5.3 times the background phosphorus loss. Litterature quantifying the increase in losses of suspended sediment during harvest were sparse. There may be incidental losses due to heavy machinery for harvesting and transport of stems, but these losses of sediment have not been quantified in the literature discussed in this report.

6.3 Other effects on water quality

Other conditions in water may also change with removal of forested areas, like temperature and colour. Trees give shadow in creeks and rivers, and can reduce heating of the water, which is especially important for salmonid fishes (Pusey and Arthington 2003). Temperature also affects the vertical circulation in lakes, which may change when trees are removed and temperature and wind conditions change.

Humic acids from forests cause a brownish colour of waters, and when forested land is cultivated for agriculture the water may become clearer. However, this effect will probably be counteracted by increased suspended sediments from erosion of the cultivated soil. Increased amounts of sediments may change the river bed substrate, which can be ecologically devastating in especially salmon rivers (Levasseur et al. 2011).

Changes in local water balance may occur due to decreased evapotranspiration and infiltration. Sun et al. (2005) registered more runoff after tree harvesting from areas with high precipitation in south-western USA, presumably related to reduced evapotranspiration. Compared to Norwegian conditions, the evapotranspiration in south-western USA is much higher and hence also the effect of trees on runoff is likely to be higher. A main conclusion of a larger Norwegian programme on floods (Hydra; Eikenæs et al. 2000) was that changes in forests and forestry over decades had not had any measurable impact on the floods in the Norwegian River Glomma. The area covered by forests in this catchment had, however, not changed much since 1900, but the volume of the forest had increased with 70 %. It was assumed that this change would lead to lower floods in summer and early autumn, when the trees are growing, but this could not be detected from the investigated data. On the other hand, a more recent European survey of forests impact on floods, using 287 catchments, indicated that forests have a measurable capacity to reduce floods (EEA 2015). They found that retention of precipitation was 25 % higher in catchments with 30 % forest than in catchments with only 10 % forest. Moreover, catchments with 70 % forest had 50% more retention of water than catchments with 10% forest. Coniferous forests had a retention that was about 10 % higher than forests with broad-leaved trees.

Changes in a catchment may cause major changes in the composition of biota, both terrestrial and aquatic. The first issue will not be treated here, but it should be noted that the terrestrial biodiversity in well maintained forested riparian zones is high (Montgomery 1996; Hågvar and Bækken 2005; Young-Mathews et al., 2010). The impact of trees along rivers has been studied especially for fish conditions. Riparean trees are important both for temperature, hiding from predators, for habitats, and for provision of food (Gregory et al. 1991; Lie and Sørensen 2013). Leaves and other organic residues that fall into rivers give food for invertebrates that again are eaten by fish. Freshwater pearl mussels have development stages on the gills of salmonid fishes, and impacts on fish will therefore also affect this species (Mejdell Larsen 2012). Harding et al. (1998) investigated restoration of biological conditions after removal of riparean vegetation, and found that it could take several decades before the biology along rivers was completely restored.

7 Implementing mitigation measures when cultivating new land

The adverse effects of cultivating new land may be reduced both during the transition phase and after the agricultural production has started by implementing environmental mitigation measures.

During the transition period, losses of suspended sediment and phosphorus may be reduced by shortening the period of transformation. This can reduce the erosion and soil loss. It is important to choose a period with stable weather conditions to harvest the forest, remove stems and stumps and to use the best available technology with least effect on soil structure and erosion. Nitrogen losses were shown to be lower when whole trees, including stems and branches, were removed than when these were left in piles. This can imply that new cultivation, where no piles are left, can give less nitrogen runoff than with forestry operations, but this will also depend on the extent of the land area cleared.

Open drainage ditches may contribute significantly to concentrations of suspended sediments and nutrients in streams. Constructing thresholds in ditches can trap sediments and prevent downstream impaired water quality. Additionally, filter material in ditches could be considered, to retain dissolved nutrients.

Constructed wetlands and sedimentation ponds at the drainage outlet of a cleared area contribute to reduced impact in the downstream recipient water. A review paper on this type of measure has recently been accepted for publication, and may give valuable examples for Norwegian cases where new land is cultivated (Carstensen et al. in press).

In forestry, as well as in cultivated land, it is important to leave a zone with natural vegetation along open channels and streams (Blankenberg et al. 2017). This is a requirement of the Norwegian Programme for the Endorsement of Forest Certification (PEFC) Forest Standard (2016) for all water bodies which are unlikely to run dry. If the forest is mainly consisting of large coniferous trees, the trees may fall when no longer protected from wind or mechanically supported by the other trees in the forest. In such cases it should be required to plant suitable trees in the riparian zone, using local tree species. Black and grey alder, different species of *Salix*, and rowan trees are usually well adapted to life along Norwegian rivers (Skarbøvik et al. 2018).

In agricultural production systems there are several ways to reduce losses of nutrients and sediments. A comprehensive overview of such mitigation measures is given in the web-based guidance on measures in agriculture (). These consist of measures to reduce erosion losses in the field (e.g., environmentally-friendly tilling techniques, cover crops and grass-covered waterways), and measures to prevent eroded soil from entering the water courses (buffer zones) or being transported further downstream in the catchment system once they have entered the streams (constructed wetlands, sedimentation ponds).

The lowest losses of suspended sediments and nutrients derive from grasslands. However, there are large variations depending on the intensity in livestock production, including intensity in grazing, the access of grazing animals to the stream and the amount of manure applied to these areas. Higher losses are found in cereal fields, whereas the highest are from areas with vegetable and potato production (Bechmann et al. 2017).

High intensity production has a higher risk of causing adverse effects than low intensity. Limiting the intensity of production will therefore reduce the risk of downstream pollution. The intensity may include both choice of production system, use of manure and fertiliser, use of pesticides and livestock density.

8 Conclusions

Cultivation of new land contributes to increased erosion and nutrient losses and adverse effects on water quality due to the substitution of forest and outfields by agricultural land. This literature review has shown that:

- Losses of nitrogen from agricultural areas in Norwegian studies were on average 17 times those of areas with forests and outfields. The catchments with extensive livestock (Naurstad and Volbu) were the agricultural catchments with the lowest nitrogen losses, but still on average ten times that of forested areas.
- The losses of phosphorus from agricultural areas in Norwegian studies were on average 56 times those of areas with forests and outfields. The catchments with potatoes and vegetables (Vasshaglona and Heia) were among the catchments with the highest phosphorus losses. The high losses in Vasshaglona can be explained by high phosphorus levels in the soil due to large phosphorus surplus in production, high-intensity soil tillage in connection with potato and vegetable production, as well as high rainfall and high runoff. Many of the Norwegian agricultural catchments have high runoff in combination with high concentrations of total phosphorus and dissolved reactive phosphorus in stream water, which results in large losses
- The losses of suspended sediments from agricultural areas in Norwegian studies were on average 106 times those of areas with forests and outfields. Losses of suspended sediment, phosphorus and nitrogen were on average higher in Norway compared to other Nordic countries. The highest losses of sediments are expected to derive from fields with vegetables and potatoes, followed by cereals and finally grasslands.
- Changing forests to agricultural land may also affect the hydrological cycle, water temperature, and light conditions, and therefore affect the composition of biota. Loss of biodiversity on land and in water is very likely.

The figures in the report cover a wide variation and represent a number of case studies. Thus, they cannot be considered representative and should not be used as an accurate measure of the effect of cultivating new land. Locally, the soil types and climate influences the actual losses from an area. However, the huge differences in nutrient and sediment losses from the two land use systems, and the fact that there is a marked difference even between the highest losses from forest and the lowest losses from agriculture, reveal that cultivation of new land will very likely cause long-term changes in the nutrient and sediment state in the adjoining water bodies.

Shorter term impacts are related to the process of new cultivation, where tree harvesting and seedbed preparation, which might give high nutrient and sediment losses:

- Soil solution chemistry in forest soils has been shown to be affected by harvesting, with a change in the soil organic carbon and nitrogen balance caused by a sudden input of logging residues, with increased concentrations in soil solution and increased risk of leaching of nitrogen and base cations as important effects.
- Under various weather conditions, heavy machinery used for forest harvesting of any type may damage the soil and create compaction and the risk for erosion in the wheel tracks.
- An additional effect of losses of organic matter is the complexation of heavy metals. They may be mobilised together with organic matter and leach into the recipient water body. However, heavy metal concentrations in Norwegian forests are generally low.

When new land is cultivated, environmental mitigation measures should be implemented. Open drainage ditches should be constructed with thresholds to trap sediments, and filter material to retain dissolved nutrients should be considered. Constructed wetlands and sedimentation ponds should be established and maintained at the drainage outlet of the cleared area. A zone with natural vegetation should be left along open channels and streams, and planting of additional broad-leaved trees should be considered. In the new agricultural land, appropriate mitigation measures should be maintained, as described in the Norwegian web-based guidance on mitigation measures in agriculture (www.nibio.no/tiltak).

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Appendix 1

The table below gives the average losses of total nitrogen (Tot N), total phosphorus (Tot P), and suspended solids (SS) in catchments of different land uses.

Catchment	Region/County	Period	Tot N kg/daa/year	Tot P g/daa/year	SS kg/daa/year	Original source	Discussed in source
Norway							
Storgama	Vestfold and Telemark	1986-2005	0.1	-	-	De Wit and Wright, 2008	-
Storgama	Vestfold and Telemark	2010-2017	0.297	4			
Birkenes		2010-2017	0.391	4.7			
Langtjern		2010-2017	0.196	3.8			
Kårvatn		2010-2017	0.108	3.6		Garmo and Skancke, 2018/Schartau et al. 2006	
Dalelv		2010-2017	0.068	2			
Øygardsbekken		2010-2017	0.584	6.2			
Skjeverbekken	Oslo		0.16	3	0.25	Holtan&Holtan, 1993	Bækken&Bratli 1995
Andebu I		1972-1975	0.11	0.7			
Andebu I	Vestfold	1975-1977	0.14	6		Haveraaen, 1981	Bækken&Bratli 1995
Andebu II, før hogst		1972-1975	0.14	1.9			
Coniferous forest				4.6	0.54		
Mixed forest I				3.4	0.38		
Mixed forest II	Telemark	1975-1979		5.6	1.04	Rognerud, Berge & Johannesen, 1979	Bækken&Bratli 1995
Mixed forest III				6.2	0.53		
Mixed forest IV				6.2	0.71		
Svela	Rogaland	1993-1994	0.589	2.6		Kaste et al., 1995	Bækken&Bratli 1995
Høgmo	Rogaland	1993-1994	0.238	3		Kaste et al., 1995	Bækken&Bratli 1995
Dal		1992-1994	0.925			Høyas et al., 1997;	Bækken&Bratli 1995
		1992-1998	1.03	6.45		Bechmann, 1999	Vandsemb, 2006
Svartbekken	Vestfold	1992-1994	0.32			Høyas et al., 1997;	Bækken&Bratli 1995
		1992-1996	0.23	0.31		Bechmann, 1999	Vandsemb, 2006

Tuften		1992-1994	0.28			Høyas et al., 1997;	Bækken&Bratli 1995
		1992-1995	0.31			Bechmann, 1999	Vandsemb, 2006
Gjerstad St1			0.364	4.8			
Gjerstad St4	Aust-Agder		0.338	4		Hindar pers. comm.	Bækken&Bratli 1995
Gjerstad St10			0.235	2			
Nyghagabråtan	Oppland	1993-2004	0.18	3.7	1.2	JOVA database	Vandsemb, 2006
Bjørnebekk - forest		1984-1985	0.46	13.9	2.2		
Holt - forest	Akershus	1984-1985	0.18	7.1	1.5	Lundekvam, 1986	Vandsemb, 2006
Hvamseter - forest		1984-1985	0.2	4	1.1		
Siljan - forest	Telemark	1970-1978	0.065	9	xx	Lundekvam, 1983	Vandsemb, 2006
Rakkestad - forest/lake		1977-1979	0.194	8	xx	Lundekvam, 1983	Vandsemb, 2006
Rakkestad - forest	Østfold	1977-1979	0.246	12	xx	Lundekvam, 1983	Vandsemb, 2006
Rakkestad - forest/lake		1977-1979	0.23	9.3	xx	Lundekvam, 1984	Vandsemb, 2006
Vasshaglona – forest	Aust-Agder	1994	1.5	xx	xx	Guttormsen, Lillemo, 1995	Vandsemb, 2006
Finland							
Finland, "background"			0.185	11		Mussaari, 1977	Bækken&Bratli 1995
Finland, 1 site				0.02	0.02	Kohonen, 1982 (in Ahl 1988)	Bækken&Bratli 1995
Finland, 8 sites		1981-1985	0.253	11.6		Rekolainen, 1989	Bækken&Bratli 1995
Murtopuro, before harvesting		1972-1982	0.204	14.6	0.4		
Kivipuro, before harvesting		1972-1982	0.157	6.9	0.4	Ahtiainen, 1992	Bækken&Bratli 1995
Liuhapuro		1979-1988	0.208	8.9	0.3		
Sweden							
Reivo, 85-87		1985-1987	0.102				
Ammarnäs, 85-87	Bottenvika	1985-1987	0.1795			Kvarnäs, 1989	Bækken&Bratli 1995
Moddus, 85-87		1985-1987	0.166				
Kullarna.ref.		1977-1980	0.0585			Rosén, 1982	Bækken&Bratli 1995
Kullarna.ref.		1-5 years later	0.128			Rosén, 1987	Bækken&Bratli 1995
Kullarna, before harvesting	Sea of Bothnia	1977-1980	0.0915				
Sniptjern		1977-1980	0.092			Rosén, 1982	Bækken&Bratli 1995

Tandovala			0.134		Kvarnäs, 1989	Bækken&Bratli 1995
Stenbitshöjden			0.089			
Dantersta		1978-1988	0.158		Ulén, 1989	Bækken&Bratli 1995
Dantarsta Alder mire		1978-1988	0.229		Ulén, 1989	Bækken&Bratli 1995
Kloten	Baltic Sea	1970-1977	0.106		Grip, 1982	Bækken&Bratli 1995
Anboda		1986-1986	0.1645		Westling & Hultberg, 1989	Bækken&Bratli 1995
Tiveden		1985-1987	0.1035		Kvarnäs, 1989	Bækken&Bratli 1995
Norra Kvill		1985-1987	0.079			
Torvbraten, before ditching		1980-1981	0.39	10	Lundin & Bergquist, 1990	Bækken&Bratli 1995
Halland, Tørnnersa	Kattegat	1986-1987	0.43		Fleischer et al., 1989	Bækken&Bratli 1995
Svartedalen		1985-1987	0.165		Kvarnäs 1989	Bækken&Bratli 1995
Boaberg ø		1985-1987	0.194			
Boaberg n		1985-1987	0.181			
Tresticklan	Skagerrak	1985-1987	0.224		Westling & Hultberg, 1989	Bækken&Bratli 1995
Gårdsjön		1984-1986	0.1405			
5 skogfelt		1965-1972	0.13	6	Ahl & Odén, 1974	Bækken&Bratli 1995
Midt-Sweden 3 cases	Sweden	1977-1980	0.1	4	Rosén, 1982	Bækken&Bratli 1995
Sweden	Unspecified			6.5	Brink & Gustavson, 1972	Bækken&Bratli 1995
Sweden	areas			4	Brink, 1972	Bækken&Bratli 1995
Coniferous forests			0.125		Löfgren, 1991 (NMR)	Bækken&Bratli 1995
Other countries						
Purukohu, pinepl., N. Z.			0.131	9.5	Cooper & Thomsen, 1988	Bækken&Bratli 1995
Purukohu, natural , N. Z.			0.367	12		
Canada				5	Dillon & Kirchner, 1974	Bækken&Bratli 1995
Canada /USA				12	Likens & Borman, 1974	Bækken&Bratli 1995
USA			0.06	4	Uttormark et al., 1974 (EPA)	Bækken&Bratli 1995
Wisconsin			0.372	11.2	Clesceceri et al., 1986	Bækken&Bratli 1995
East USA			0.44	8.3	Omernik, 1976	Bækken&Bratli 1995
West USA			0.311	12.4	Omernik, 1977	Bækken&Bratli 1995
Alaska			0.14	20	Stednick, 1981	Bækken&Bratli 1995

Norsk institutt for bioøkonomi (NIBIO) ble opprettet 1. juli 2015 som en fusjon av Bioforsk, Norsk institutt for landbruksøkonomisk forskning (NILF) og Norsk institutt for skog og landskap. Bioøkonomi baserer seg på utnyttelse og forvaltning av biologiske ressurser fra jord og hav, fremfor en fossil økonomi som er basert på kull, olje og gass. NIBIO skal være nasjonalt ledende for utvikling av kunnskap om bioøkonomi.

Gjennom forskning og kunnskapsproduksjon skal instituttet bidra til matsikkerhet, bærekraftig ressursforvaltning, innovasjon og verdiskaping innenfor verdikjedene for mat, skog og andre biobaserte næringer. Instituttet skal levere forskning, forvaltningsstøtte og kunnskap til anvendelse i nasjonal beredskap, forvaltning, næringsliv og samfunnet for øvrig.

NIBIO er eid av Landbruks- og matdepartementet som et forvaltningsorgan med særskilte fullmakter og eget styre. Hovedkontoret er på Ås. Instituttet har flere regionale enheter og et avdelingskontor i Oslo.

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