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Degradation and mobility of pesticides in Norwegian soils

Opinion of the Panel on (or the Scientific Committee) of the Norwegian Scientific Committee for Food Safety

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Opinion of the Panel on Plant Protection Products of the Norwegian Scientific Committee for Food Safety
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Assessed and approved

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Summary

In this report the following topic of pesticides and fate in Norway has been outlined covering: 1. Factors influencing degradation of pesticides. 2. Description and update of datasets on soil and climate in agricultural areas. 3. Normalization of field degradation data as input for modelling fate. 4. Use of degradation data from Norway in model scenarios.

Norwegian laboratory degradation studies indicate that increased soil organic carbon content enhances degradation rates of pesticides that show low sorption (e.g. metalaxyl, bentazone) ,due to increased microbial activity. Whereas pesticides that sorb moderately to strongly to soil (e.g. boscalid, propiconazole), display reduced degradation as organic carbon increases as a consequence of sorption and reduced bioavailability.

Recent DegT50 field studies display a large variation in fungicide degradation rates from Klepp in the south to Tromsø in the north. For the mobile herbicide bentazone, no effect of climate was observed, as degradation rates were coherent at all sites, probably due to rapid leaching. The climate (temperature) seems to be more determinate for fungicide degradation rates than the soil type. Fungicide degradation was slow at two northern sites having low soil temperatures, even though microbial biomass was hugely different at the sites. How soil temperature and moisture affects microbial activity and diversity in various soils, climates and crops is important for the understanding of degradation capacity in Norwegian soils and fields. Microbial activity could be related to both soil, climate and crops/cropping regime – as well as to the nature of the soil organic matter.

The fact that DegT50 values are very much shorter than laboratory values at the same reference conditions, may point to some systematic error in the normalization procedure (e.g. the default simplifications in the Walker and Arrhenius equations), or that the parameters affecting degradation in the laboratory are different from the parameters that affect degradation in the field. Consequently, lab-derived and field-derived DegT50matrix values should be compared and interpreted with care.

The large variations in normalized DegT50 values obtained in field studies in Norway as well as in other regions in Norway cannot be explained by differences in the associated parameters characterizing the soil and microbial community. It is therefore not possible to determine if a certain field study is more or less representative for "Norwegian conditions". As a conservative approach, the highest, normalized DegT50 from the European field studies should be selected for the Norwegian risk assessment independent on geographic vicinity. As an alternative, when a sufficient number of data are available, a high percentile (e.g 80 or 90-percentile) should be used rather than the geomean.

Each agricultural region in Norway is dominated by one specific soil type for each region. Albeluvisol, Cambisol, Umbrisol, Stagnosol and Histosol in respectively Eastern Norway south, Eastern Norway north, Rogaland, Trøndelag and North of Norway. New updates for Norway

include especially Umbrisols and Histosols rich in organic matter. Albeluvisols, Cambisols and Stagnosols are representing the main soil types in the agricultural area in Norway. These are also included in the groundwater (Rustad and Heia) and surface water scenarios (Syverud) developed for Norway. Experience from pesticide fate in the organic rich soils on the south west coast and north of Norway is limited.

Compared to the "normal" temperature and precipitation from 1961 to 1990 with a "new normal" from 1991 to 2014, the climate has changed. For the five described agricultural areas in Norway, annual temperature has increased in average 1 degrees for all five regions and seasons for the new normal. The rainfall has increased for all seasons and regions except for the Northern Norway (Holt in Tromsø) and summer season at Kvithamar (Trøndelag) with lower precipitation in June to September. Annually the precipitation has increased approximately 100 mm in average.

The existing Norwegian scenarios in groundwater and surface water seem to be representative in the meaning of covering the main soil types in the central agricultural areas in South Eastern Norway. However there are no scenarios covering areas of South West and North of Norway containing soil with high organic content, slow degradation and heavy rainfall. Vulnerable areas are not included in these scenarios as the idea of the representativity of soil was to include the main soil types covering the most of the agricultural production areas. The vulnerable areas deals with smaller areas and has to be treated separately. Vulnerable areas are areas with high groundwater levels and sandy soil and mobile pesticides. Hilly areas with clay soil represent high risk of surface runoff with strongly sorbed pesticides. We are lacking experience from areas with high content of organic matter causing slow degradation, combined with heavy rainfall.

A database with representative soils and climates for various crops should be established in Norway and utilized in a targeted risk assessment approach. Then, the degradation of pesticides to be used in for example fruit/berry cropping, could be evaluated in respect to representative and vulnerable soils and climates in fruit/berry regions in Norway.

A correct risk assessment of pesticide degradation in Norwegian agricultural soils should take the varying climatic zones, the diversity in agricultural soils and crops in Norway into consideration before formulated pesticides are approved. Risk assessment should be based on soils and climates most prevalent for the crop to which the pesticide is to be applied, in addition, vulnerable areas with slow degradation and/or high leaching/runoff risk should be recognized.

Key words: VKM, Norwegian Scientific Committee for Food Safety, Norwegian Food Safety Authority, pesticides, plant protection products, mobility, degradation, field studies, Norwegian conditions

Sammendrag på norsk

I denne rapporten har følgende tema om plantevernmidler og skjebne i Norge vært behandlet: 1. Faktorer som påvirker nedbrytningen av plantevernmidler 2. Beskrivelse og oppdatering av datasett på jordsmonn og klima i jordbruksområder 3. Normalisering av nedbrytingsdata fra feltforsøk som input for modellering av plantevernmidler skjebne i naturen 4. Bruk av nedbrytingsdata fra Norge i modellscenarier.

Norske nedbrytingsstudier i laboratorium indikerer at nedbrytningshastigheten av nøytrale plantevernmidler som viser lav absorpsjon øker med økende innhold av organisk materiale i jorda, på grunn av økt mikrobiell aktivitet. Plantevernmidler som bindes moderat til sterkt til jord, viser redusert nedbrytning med økende organisk karbon som følge av økt binding og redusert bio-tilgjengelighet.

Nyere nedbrytingsstudier (DegT50) i felt fra Klepp i sør til Tromsø i nord viser en stor variasjon i nedbrytningshastighet for soppmidler. For det mobile ugressmiddelet bentazon, ble det ikke påvist effekt av klima. Klimaet (temperatur) synes å ha større betydning for nedbrytningshastigheten enn jordtype. Nedbrytingen av soppmidlene var sakte på to steder med lave jordtemperaturer, selv om mikrobiell biomasse var forskjellig. Nedbrytningshastigheten (DegT50) i felt er mye kortere enn laboratorieverdier ved de samme betingelser noe som kan tyde på systematisk feil i normaliseringsprosedyren (F.eks. standard forenklinger i Walker og Arrhenius ligninger), eller at de parametere som påvirker nedbrytning i laboratoriet er forskjellige fra parameterne som påvirker nedbrytning i felt. Følgelig bør lab-avledet og felt avledet $\text{DegT50}_{\text{matrix}}$ verdier sammenlignes og tolkes med forsiktighet.

Hver landbruksregion i Norge domineres av én bestemt jordtype for hver region. Albeluvisol, Cambisol, Umbrisol, Stagnosol og Histosol i henholdsvis Øst-Norge sør, Øst-Norge nord, Rogaland, Trøndelag og Nord-Norge. Nye oppdateringer for Norge omfatter spesielt Umbrisols og Histosols som er rik på organisk materiale. Albeluvisols, Cambisols og Stagnosols representerer de viktigste jordtyper i jordbruksarealet i Norge. Disse er også inkludert i grunnvann (Rustad og Heia) og overvann scenarier (Syverud) utviklet for Norge. Erfaring fra plantevernmiddel-skjebne i organisk jordsmonn på sørvestkysten og Nord-Norge er begrenset

Sammenlignet med "normal" temperatur og nedbør 1961-1990 med en "ny normal" fra 1991 til 2014, har klimaet endret seg. For de fem beskrevne jordbruksområdene i Norge har årlig temperatur økt i gjennomsnitt 1 grad for alle fem regioner og årstider for den nye normalen. Nedbøren har økt for alle årstider og regioner bortsett fra Nord-Norge (Holt i Tromsø) og sommersesongen på Kvithamar (Trøndelag) med lavere nedbør i juni til september. Årlig nedbør har økt ca. 100 mm i gjennomsnitt.

De eksisterende norske scenarier i grunnvann og overflatevann synes å være representative i betydningen av å dekke de viktigste jordtyper i de sentrale jordbruksområder i Sør-Øst-Norge. Men det er ingen scenarier som dekker områder av Sør-Vest og Nord-Norge som inneholder jord med høyt organisk innhold, langsom nedbrytning og kraftig regn. De sårbare områdene omhandler mindre områder og bør behandles separat. Sårbare områder er områder med høye grunnvannsnivå og sandholdig jord og mobile plantevernmidler. Kuperte områder med leirjord representerer høy risiko for overflateavrenning med sterk binding av plantevernmidler. Vi mangler erfaring fra områder med høyt innhold av organisk materiale som forårsaker langsom nedbrytning i kombinasjon med kraftig regn.

En database med representative jordsmonn og klima knyttet til ulike dyrkingspraksis bør etableres i Norge og benyttes i en målrettet risikovurdering. Nedbrytning av plantevernmidler i for eksempel frukt / bær, kan bli vurdert i forhold til representative og sårbare jordsmonn og klima for regioner i Norge.

En riktig risikovurdering av plantevernmidlers nedbrytning i norsk landbruksjord bør ta varierende klimatiske soner, mangfoldet i jordbruksjord og avlinger i Norge i betraktning før formulert plantevernmidler er godkjent. Risikovurdering bør være basert på jordsmonn og klima knyttet til de områdene med størst jordbruksaktivitet, men i tillegg bør sårbare områder med langsom nedbrytning og / eller høy utlekking / avrenning risiko lokaliseres.

Background and terms of reference as provided by the Norwegian Food Safety Authority

This report is written on request from the Norwegian Scientific Committee for Food Safety ordered by Norwegian Food Safety Authority (appendix A). Background for the requested report is that approval of new pesticides follows standardized procedures and guidelines for laboratory and field studies in EU when evaluating degradation and exposure of pesticides. The main question from the Norwegian Food safety Authority is: Are these studies relevant to be used and representing Norwegian condition?

In this report, the following four questions from Norwegian Food Safety Authorization will be investigated:

1. What are the most important factors contributing to the degradation of pesticides under Norwegian conditions, and which are negligible and others specific to Norwegian conditions?

2. The existing dataset on normal Norwegian soil and climate parameters need to be updated. What is the normal span in soil and climatic parameters in Norwegian agricultural today – applicable for pesticide risk assessment?

3. Can normalized pesticide degradation data (= $\mathrm{DegT50_{matrix}}$) from European sites be applied as input parameters for modelling the fate of pesticides in Norway?

4. Can pesticide degradation data from European sites be used as input in pesticide modelling if the soil and climate parameters at the sites are exceeding the normal Norwegian span? How much exceedances can be accepted?

1 Introduction

Norway has recently adopted the "Sustainable pesticide use directive" (SUD), Directive 2009/128/EC and Regulation EC 1107/2009. To reduce the workload and time spent on authorization of pesticide formulations, the principle of zonal evaluation (Figure 1-1) and mutual recognition is embedded in Regulation EC 1107/2009 for the placing of plant protection products on the market. The Northern Zone co-operation, as outlined in the 'Guidance document on work-sharing in the Northern zone in the authorization of plant protection products, include the EU member states Denmark, Sweden, Finland, Estonia, Latvia, and Lithuania, as well as the EEC/EFTA members Norway and Iceland. However, the three European pesticide authorization zones are very broad as compared to the various climatic zones in Europe; e.g. Norway has 3-4 different climatic zones (Figure 1-2), not reflected well by the climate neither in the other Nordic-Baltic countries nor in central and southern EU. But how well does pesticide degradation data from middle and southern EU relate to degradation rates in Norwegian climatic conditions? Which are the key determining factors for pesticide degradation in Norway?

Figure 1-1 EU zones for the authorization of pesticides. Norway is included in the Northern zone, also called the Nordic-Baltic zone.

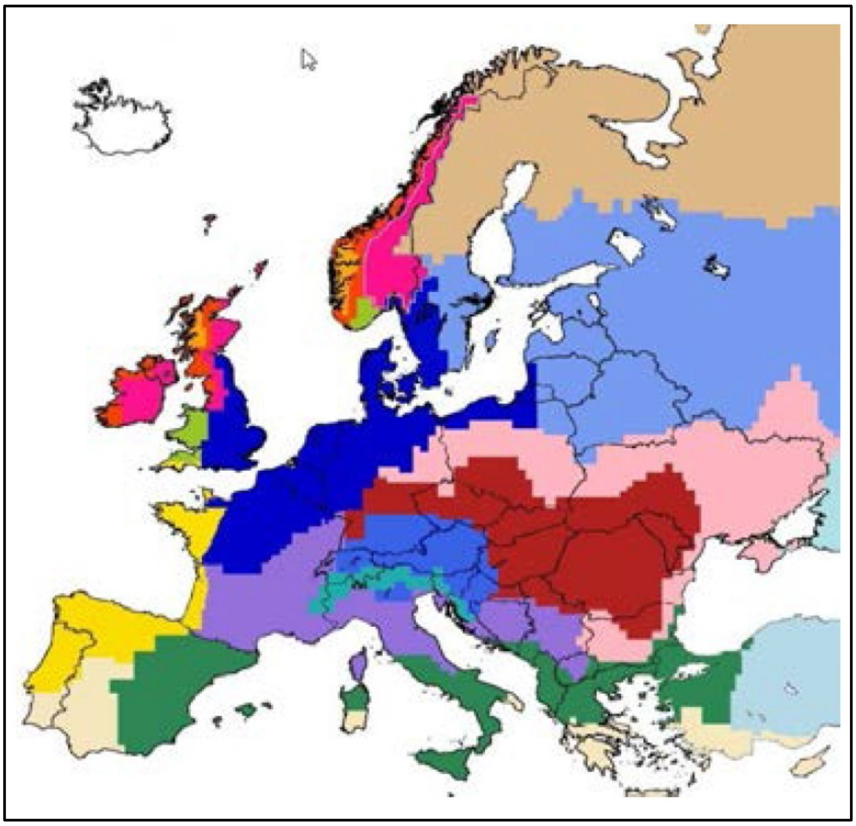


Figure 1-2 Climatic zones for pesticide modelling (Blenkinsop et al., 2008).

2 The factors most affecting pesticide degradation rates in Norway

This part of the report deals with the parameters that affect pesticide degradation rates in Norwegian soils, namely:

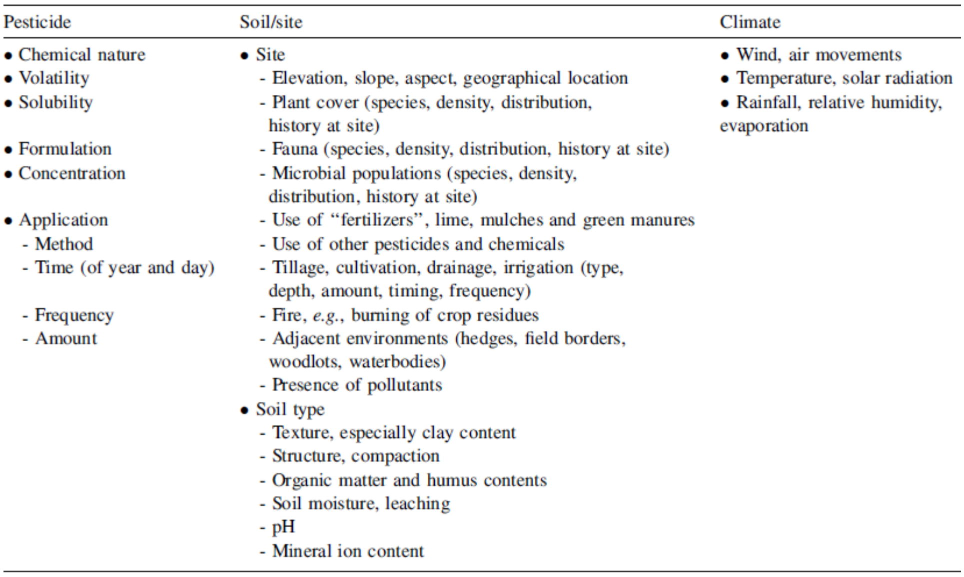
1) The characteristics of the pesticide itself

2) The physio-chemical and biological properties of the soil

3) The climate

There are quite a lot of factors that influence the degradation or persistence of pesticides in soils, see a list in Table 2-1. We will discuss the most important drivers for pesticide degradation in Norwegian soils and climate and arrive at a conclusion regarding their relative importance (Chapter 2.5).

Table 2-1 Some factors influencing the persistence of pesticides in soil (Arias-Estévez et al., 2008).



Pesticide	Soil/site	Climate
• Chemical nature	• Site	• Wind, air movements
• Volatility	- Elevation, slope, aspect, geographical location	• Temperature, solar radiation
• Solubility	- Plant cover (species, density, distribution, history at site)	• Rainfall, relative humidity, evaporation
• Formulation	- Fauna (species, density, distribution, history at site)	
• Concentration	- Microbial populations (species, density, distribution, history at site)	
• Application	- Use of "fertilizers", lime, mulches and green manures	
- Method	- Use of other pesticides and chemicals	
- Time (of year and day)	- Tillage, cultivation, drainage, irrigation (type, depth, amount, timing, frequency)	
- Frequency	- Fire, *e.g.*, burning of crop residues	
- Amount	- Adjacent environments (hedges, field borders, woodlots, waterbodies)	
	- Presence of pollutants	
	• Soil type	
	- Texture, especially clay content	
	- Structure, compaction	
	- Organic matter and humus contents	
	- Soil moisture, leaching	
	- pH	
	- Mineral ion content	

2.1 Definition of DT50 vs. DegT50 and their use in risk assessment

In traditional terrestrial field dissipation studies, the dissipation of a pesticide in the field is studied as a response to all normal processes at the field, including plant uptake, photo degradation, evaporation, soil management, leaching etc. This gives an estimate of the pesticide dissipation half-life (DT50, with D for dissipation); influenced by transformation and transport under representative actual use conditions. In the laboratory, degradation half-lives can be found in a controlled environment. Degradation half-lives determined at reference conditions (soil temperature 20°C and field capacity pF2) are called $DegT50_{matrix}$.

However, a need has been identified to produce field degradation data that better could be compared to the laboratory degradation values. Hence, field degradation study protocols were established, where all processes, except the formation of degradation products by chemical and microbial processes, were minimized. Such field degradation studies produce DegT50 values. Losses from e.g. plant uptake and surface processes like photo degradation and evaporation are eliminated by mixing the pesticide into the topsoil shortly after application and incubating in a no crop field. It is then possible to derive $DegT50_{matrix}$ values from field degradation studies through a normalization procedure (see Chapter 5), where seasonal field temperatures and moisture are normalized to 20°C and field capacity. In general, field-derived $DegT50_{matrix}$ will be lower than lab-derived $DegT50_{matrix}$ values and whether to use lab-derived or field-derived $DegT50_{matrix}$ values, or a combination, as model input parameters are further explained in EFSA guideline no. 12(5), 2014 (EFSA, 2014).

Pesticide DegT50 values are critical input values in models assessing soil, surface water and groundwater pesticide exposure levels. Normally a geo-mean DegT50 is calculated from several field studies carried out at different locations. The models require that half-life values should represent degradation within the soil matrix exclusively (and at reference conditions); hence $DegT50_{matrix}$ values are preferred input values. It may be possible to derive DegT50 values from "old" dissipation (DT50) studies, by a procedure described in the EFSA guidance (EFSA, 2014).

The $DegT50_{matrix}$ values represent conservative endpoints for comparison against the field persistence criteria in the European pesticide legislation, but they may be used in comparison against the POP, PBT and vPvB criteria. DT50 values should be used for the assessment of field persistence criteria (EFSA, 2014).

2.2 Pesticide properties that affect degradation rate

Pesticide properties that affect degradation rate are molecular size, molecular structure, chemical functional groups, solubility, polarity and charge distribution of interacting species, and the acid–base nature of the molecule. Water solubility (Sw), polarity (Kow) and acid-basic character (pKa) are among the most important properties affecting the sorption behaviour, and hence degradation, of pesticides in soils (Hiller et al., 2008; Nemeth-Konda et al., 2002). The sorption of pesticides in soils within a common pH-range (4-8) usually decreases according to the following:

a) Pesticides with low water solubility > pesticides with high water solubility

b) Non-polar pesticides > polar pesticides

c) Neutral pesticides > basic pesticides > acidic pesticides

The molecular feature of the pesticide strongly affects its sorption and degradation (Figure 2.2-1).

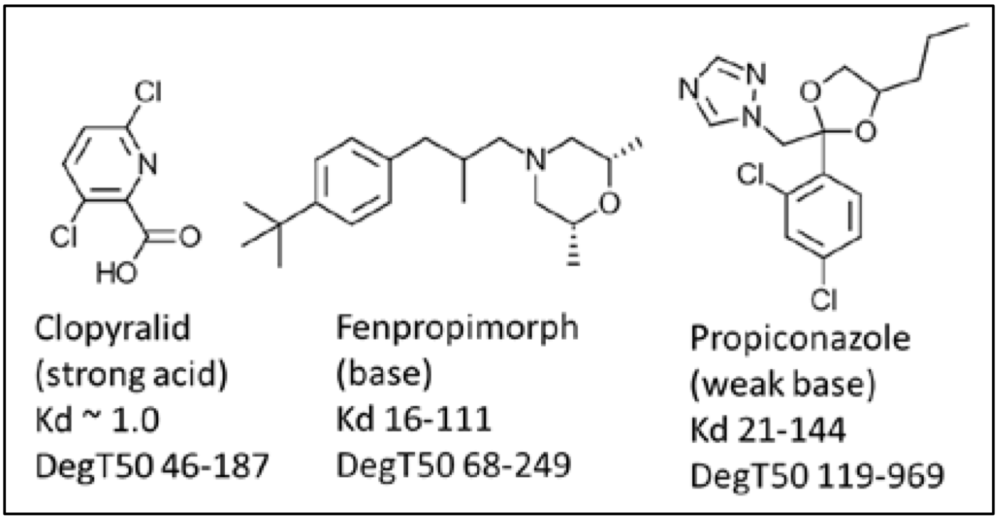


Figure 2.2-1 Examples of molecular structures of pesticides.

Polar groups (e.g. OH, COOH, NH_2) provide microorganisms a site of attack, whereas halogen substituents (e.g. F and Cl) or alkyl substituents ($-(CH_2)_n$) render the molecule more resistant to biodegradation (Cork and Krueger, 1991). Molecular features also render some pesticides more prone to abiotic degradation by hydrolysis and/or photolysis than others. Hydrolytic reactions can be acid- or base-catalysed (Chaplain et al., 2011).

2.3 Soil parameters that affect pesticide degradation rate

2.3.1 Organic matter/carbon

Soil organic matter is usually measured by determining the amount of organic carbon present, using digestion or combustion techniques (Wauchope et al., 2002). The content of organic matter or organic carbon (OC) in the soil has a direct influence on the microbial biomass in the soil and hence the soil microbial capacity to degrade pesticides. A strong positive correlation between OC content and pesticide degradation rates have been demonstrated in laboratory degradation studies (Kah et al., 2007). On the other hand, the soil's capacity to sorb pesticides tends to be larger in soils with higher OC, and a strong sorption is generally assumed to reduce bioavailability of the pesticides and decrease degradation. A simple link between sorption capacity and degradation rate has proven difficult to obtain, as sorption-degradation relationships depend on the pesticide itself as well as other soil parameters (Kah and Brown, 2007). In the field, leaching of the pesticide impedes the interpretation of degradation, so that the disappearance of the pesticide in field studies where leaching is not controlled, must be interpreted as dissipation, not degradation. Factors that affect degradation rates may better be studied in laboratory degradation studies. DegT50 values from laboratory degradation studies performed in Norwegian soils at standard conditions (e.g. soil temperature of 20°C and water holding capacities of 50-70%) are presented in Table 2.3.1-1. The results can be summarized as follows: For pesticides that show low sorption (e.g. metalaxyl, increases in organic carbon in the soils may enhance degradation due to increased microbial activity, whereas for pesticides that sorb moderately to strongly to soil (e.g. boscalid, propiconazole), degradation is reduced as organic carbon increases due to sorption and reduced bioavailability. These trends are easier to observe in degradation studies with fairly similar soils but with a wide range of OC content, but may not apply to all pesticides and soils, especially not pesticides that can be degraded chemically or whose sorption is pH-dependent.

The slow degradation of propiconazole in a loam from Vollebekk (Ås), cannot be explained by a high content of organic matter (Table 2.3.1-1), and the impact of other factors such as aged sorption and microbial diversity must be considered. Also, across a field, organic matter may not be uniformly distributed. There may be large spatial variations in the soil capacity to degrade pesticides, as observed in a small sloping silt loam site at Grue (Table 2.3.1-1). Degradation of the moderately strong sorbing propiconazole was twice as fast in soils from the top and middle of the slope of the field as compared to soils from the bottom of the slope. The mobile herbicide clopyralid was degraded three times more rapidly in soil from the top and bottom of the slope as compared to soil from the middle part of the slope (all at 20°C in laboratory). Large horizontal variability in glyphosate mineralization (coefficient of variation: 37%; cf. Table 2.3.4-1) has also been shown in lab scale studies with soil from the same field site (Stenrød et al., 2006). In this case the variability was linked to micro-topography, showing faster glyphosate degradation in the depressions. Spatial variability in soil texture and OC content, as a function of topographic position; with higher OC content in terrain depression as compared to the hilltops and slopes (Jarvis et al. 2001) could be one explanation. Furthermore, the microbial activity may be higher in lower slopes due to accumulation of nutrients and moisture (Florinsky et al., 2004), and the hilltops may have more macro-pore flow (due to higher clay content) as compared to the hollows (Roulier and Jarvis, 2003), which add to the spatial variability in pesticide degradation as well as leaching of pesticides. The interpretation of degradation rates can be quite complicated, and even more so if laboratory degradation rates are to be compared to field degradation rates.

Table 2.3.1-1 Degradation of pesticides in Norwegian soils (lab-studies, 20°C). SFO kinetics - unless otherwise stated.

Pesticide	Soil type	$DegT50_{LAB}$ (days)	Sorption (K_d)	OC (%)	pH (H_2O)	pKa	Soil texture (%)		
							Sand	**Silt**	**Clay**
Isoproturon	Silty clay loam (Rustad)	13	2.9	1.9	6.6	-	13	60	27
($DT50_{EU}$ 7-18)	Sandy loam (Heia)	13	2.2	2.2	6.4		65	30	5
Metalaxyl	Silty clay loam (Rustad)	21	1.8	1.9	6.6	0	13	60	27
($DT50_{EU}$ 14-43)	Sandy loam (Heia)	46	0.9	2.2	6.4		65	30	5

Pesticide	Soil type	DegT50$_{LAB}$ (days)	Sorption (K_d)	OC (%)	pH (H_2O)	pKa	Soil texture (%)		
	Loam (Syverud)	38	< 1.0	3.1	5.5		26	47	27
	Silty clay loam (Bjørnebekk)	107	< 1.0	1.0	6.4		7	62	31
Bentazone	Loam (Vollebekk)	32^{SFO} 17^{DFOP}		1.6		3.28	30	46	25
(DT50$_{EU}$ 8-102)	Sandy loam (Særheim)	39		3.3			57	32	11
	Sandy loam (Kvithamar)	81		1.8			45	49	7
	Sandy loam (Holt)	29		5.5			58	35	7
	Sandy loam (Hole)	133	0.09_{Kf}	1.4	6.3		49	46	5
	Loam (Kroer)	87	0.07_{Kf}	2.5	5.5		36	45	19
Boscalid	Loam (Vollebekk)	260	13	1.6		-	30	46	25
(DT50$_{EU}$ 108-384)	Sandy loam (Særheim)	510	36	3.3			57	32	11
	Sandy loam (Kvithamar)	192	14	1.8			45	49	7
	Sandy loam (Holt)	(6191)	70	5.5			58	35	7
Fenpropimorph	Loam (Vollebekk)	68^{HS}	70	1.6		6.98	30	46	25
(DT50$_{EU}$ 10-124)	Sandy loam (Særheim)	137	35	3.3			57	32	11
	Sandy loam (Kvithamar)	78^{DFOP}	16	1.8			45	49	7
	Sandy loam (Holt)	249	111	5.5			58	35	7
Propiconazole	Loam (Syverud)	281	26	3.1	5.45	1.09	26	47	27
(DT50$_{EU}$ 29-70)	Silty clay loam (Bjørnebekk)	144	21	1.0	6.4		7	62	31
	Sandy loam (Hole)	137	27_{Kf}	1.4	6.3		49	46	5
	Loam (Kroer)	210	36_{Kf}	2.5	5.5		36	45	19
	Silty clay loam (Skuterud)	172		1.9	5.8		13	60	27
	Loam (Vollebekk)	(969)	32	1.6			30	46	25
	Sandy loam (Særheim)	489	98	3.3			57	32	11
	Sandy loam (Kvithamar)	119	41	1.8			45	49	7
	Sandy loam (Holt)	703	144	5.5			58	35	7
	Silt loam (Grue) Top	184	32	0.8	5.4		42	55	4

Pesticide	Soil type	DegT50$_{LAB}$ (days)	Sorption (K$_d$)	OC (%)	pH (H$_2$O)	pKa	Soil texture (%)		
	Silt loam (Grue) Middle	177	39	0.9	5.9		39	58	4
	Silt loam (Grue) Bottom	359	48	1.0	5.7		45	52	3
Azoxystrobin	Silt loam (Grue) Top	54	9.0	0.8	5.4	-	42	55	4
(DT50$_{EU}$ 279)	Silt loam (Grue) Middle	54	9.3	0.9	5.9		39	58	4
	Silt loam (Grue) Bottom	67	8.3	1.0	5.7		45	52	3
Clopyralid	Silt loam (Grue) Top	46	None	0.8	5.4	2.01	42	55	4
(DT50$_{EU}$ 13-65)	Silt loam (Grue) Middle	187	None	0.9	5.9		39	58	4
	Silt loam (Grue) Bottom	57	None	1.0	5.7		45	52	3
Metribuzin	(Målselv)	$385^{5°C}$ $76^{15°C}$ $22^{20°C}$	0.60	0.9	6.6	0.99	64	34	3
(DT50$_{EU}$ 5-18 d.)	Silt loam (Grue)	$330^{5°C}$ $105^{15°C}$	0.36	0.9	5.9		47	49	4
Glyphosate	Clay loam (Askim)	35^{FOMC} DT90>$600^{15°C}$	88	1.0	6.7		24	53	24
(DT50$_{EU}$ 4-180)	Clay loam (Syverud)	67^{FOMC} DT90>$200.000^{15°C}$	113	2.6	5.8		27	46	26
Fluazinam	Sandy loam (Rygge)	$47^{15°C}$	453	1.1	6.5	7.34	65	27	9
(DT50$_{EU}$ 17-263)	Silt loam (Roverud)	$107^{15°C}$	320	1.4	6.5		17	73	10

2.3.1.1 The nature of organic matter and its effect on sorption

As can be seen from Table 2.3.1-1, explaining or predicting biodegradation endpoints in a given soil based on OC content and sorption may be inherently difficult. It brings us to the question:

What is "organic matter"?

Soil organic matter originates from crop residues, microbial biomass and organic amendments. It has very heterogeneous composition and contains both hydrophilic and hydrophobic groups (Calvet, 2005). "Well-decomposed organic matter forms humus, a dark brown, porous, spongy material that has a pleasant, earthy smell" (USDA, 1996). The soil organic matter is subject to a constant turnover in which microorganisms degrade organic matter (Burauel and Führ, 2000). Humic substances may be separated into recognizable compounds (polysaccharides, lignins and polypeptides) and amorphous polymers (fulvic acid, humic acid and humin) (Figure 2.3.1.-1). The fulvic and humic acids can be extracted from soil with aqueous sodium hydroxide, leaving the humin un-extracted. Acidification of the brown extract will cause humic acid to precipitate, leaving the fulvic acids in solution (McBride, 1994).

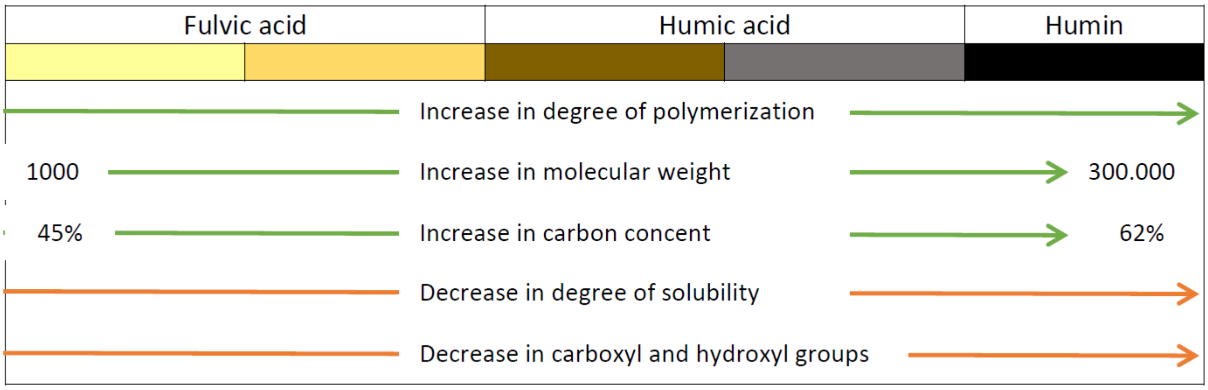


Figure 2.3.1.1-1 Soil organic matter is constituted by humus and unaltered debris. Humus can further be divided into three categories; fulvic acid, humic acid and humin, with distinctive polymeric traits.

Pesticides and/or their transformation products can form bound, non-extractable residues (NER) to humic substances. NER formation can be studied by adding radiolabelled pesticides to soil. The non-extractable residues of hydrophobic compounds are associated with the humin fraction (Ding et al., 2002; Kloskowski and Fuhr, 1987; Kohl and Rice, 1998); as observed for propiconazole (Kim et al., 2003). The nature of the organic matter influences the formation of NERs for most pesticides (EFSA PPR Panel, 2015). Furthermore, the persistence and magnitude of the formation of pesticide NER in soils are found to be mainly compound specific, as pesticides or degradation products with reactive chemical groups, such as e.g. aniline ($C_6H_5NO_2$) or phenol (C_6H_5OH), have a tendency to form high NER ratios (Barriuso et al., 2008; Burauel and Führ, 2000). But NER formation is also correlated with soil microbial activity and the amount of organic matter (Kaufman and Blake, 1973). As NERs are non-extractable, the NER proportion is not dealt with in aerobic degradation studies with non-labelled pesticide following the OECD 307 guideline. Implications from pesticide NER formation is that degradation rates of pesticides may be overestimated and their persistence underestimated.

Whereas formation of NER addresses the more or less irreversible sorption of pesticides and their transformation products, aged sorption describe the time-dependent reversible sorption of pesticides (and metabolites). Sorption is usually progressive in nature, starting with a fast initial adhesion towards accessible sorption sites followed by a slower, but still reversible, sorption for hours or days tending towards equilibrium, and finally a third phase; the aged sorption phase, with slow sorption proceeding for weeks and years with decreasing extractability. Aged sorption may be difficult to distinguish from degradation and NER formation (Wauchope et al., 2002).

Pesticides sorbed by aged sorption i.e. the non-equilibrium sorbed fraction – cannot be desorbed by water but by organic solvents. Aged sorption is directly correlated with the amount of organic matter; however, the clay and silt minerals may be governing components for ageing in soils or sub-soils low in organic matter (EFSA PPR Panel, 2015).

A two-site conceptual model of aged sorption was recently used to assess aged sorption of the fungicides propiconazole and boscalid in four Norwegian soils. Preliminary results display aged sorption in three out of the four soils (Almvik et al., unpublished data). Implications are that if only sorption coefficients determined by 24-hour batch equilibrium are used in risk assessment of pesticides that are prone to increased sorption with time, the mobility of the pesticides may be overestimated.

The humic substances in soils will vary widely in composition with impacts on sorption, aged sorption, NER formation and degradation rates. The importance of taking the chemical properties or nature of soil organic matter into account when predicting environmental behaviour of pesticides has recently been recalled in an EFSA opinion (EFSA PPR Panel, 2013). Much of the uncertainty regarding the elucidation of aged sorption and NER formation – and hence degradation - arises from our poor understanding of the structure of soil organic matter (Arias-Estévez et al., 2008; EFSA PPR Panel, 2015).

2.3.2 Minerals

Clays (as silicate minerals), oxides and hydroxides are minerals involved in the adsorption of pesticides (Calvet, 1989). The smallest particles in soil (<0.002 mm) are called clay, and clayey soils have a much larger internal reactive surface area than courser soils, providing a greater surface area for adsorption of pesticides (Edwards, 1975). The mineral surfaces are mainly hydrophilic due to hydroxyl groups and exchangeable cations. Cations associated with clay minerals are e.g. iron, magnesium, alkali metals and alkaline earths. Pesticides likely adsorb on external surfaces of clay particles rather than in inter-lamellar space and the adsorption increases with the specific surface of clays (Barriuso et al., 1994). Oxides and hydroxides are frequently associated to clays, and their charge depends on the soil pH (Calvet, 1989). Clay may be covered by iron oxides, resulting in a decreased negative charge on clay, since iron oxides are protonated and positively charged at low pH (< pH 6.5-9.5) and increasingly negative at high pH.

Pesticides are generally strongly adsorbed to soils that are high in organic matter – but also to soils high in clay minerals. In soils low in organic carbon, pesticides that normally sorb well to OC, might instead sorb to the clay mineral fraction. Some pesticides even sorb preferentially better to clay minerals than to OC, e.g. glyphosate and pesticides that are cationic.

The fungicide fenpropimorph is a base (pKa 7.0) and is a cation at a soil pH below 7 and its sorption is reported to be better correlated with clay content than with OC content (Kah and Brown, 2007). In some Norwegian soils, fenpropimorph sorbs stronger to a soil with high clay content (25% clay, Vollebekk-soil) than to soils with lower clay content (7-11% clay) (Table 2). Nevertheless, fenpropimorph sorbs even stronger to soil with a very high OC content and low clay content (Holt-soil; 5.5% OC). The strong sorption to the clay-rich Vollebekk soil was however not accompanied by a slow degradation, on the contrary, degradation of fenpropimorph in this soil was much faster than in the other soils, and displayed hockey-stick (HS) kinetics. Interestingly, at a soil temperature of 10°C, fenpropimorph degraded with single-first-order (SFO) kinetics in the Vollebekk soil (data not shown). This suggests that the microorganisms in this soil are able to degrade clay-sorbed fenpropimorph at 20°C (DT50LAB = 68 days), whereas the microorganisms in the Holt soil are not as capable to degrade OC-sorbed fenpropimorph (DT50LAB = 249 days). The bi-phasic degradation kinetics of fenpropimorph in the Vollebekk and Kvithamar soils may imply time-dependent (aged) sorption in these soils.

In conclusion: some pesticides, especially bases and polar pesticides, sorb well to clay minerals, but sorption will be dependent on the clay mineral type and composition in soils, and the effect of sorption on degradation must be evaluated on a case-by-case basis in each soil.

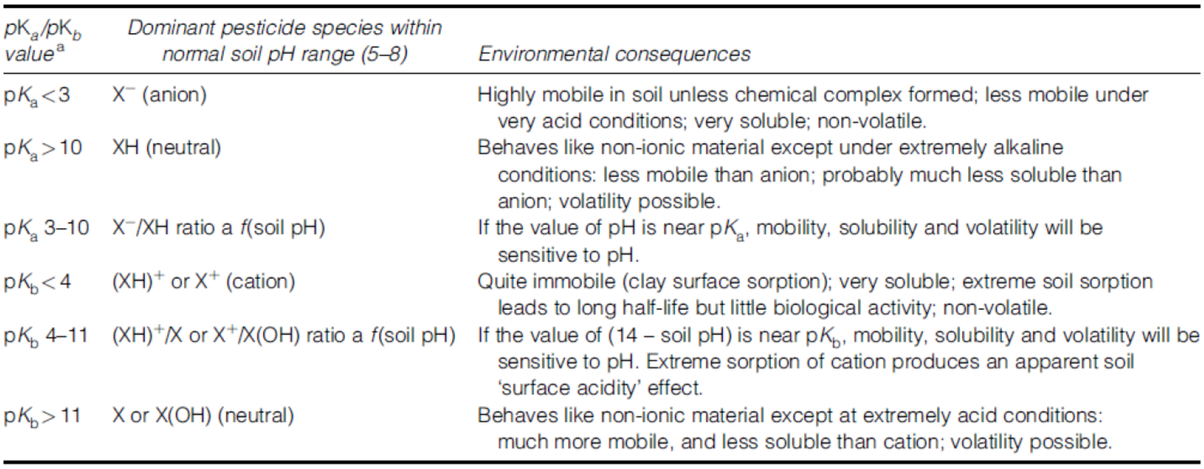
2.3.3 Soil pH

Soil pH may affect pesticide sorption, microbial abundance and chemical degradation.

Soil pH affects the rate of abiotic degradation, e.g. sulfonylurea herbicides are rapidly degraded by hydrolysis at pH < 5 (Kah and Brown, 2006). Soil pH also affects microbial activity, abundance and diversity; bacterial abundance may increase twofold from pH 4 to 8 (Rousk et al., 2010), with concomitant effects on pesticide degradation rates.

In general, soil pH changes have only minor effects on the adsorption of non-ionic molecules, but the sorption of ionisable pesticides may be highly sensitive to soil pH. Acidic pesticides (anionic groups) will have quite low Kd values because they are repelled by the negative net charge of soil surfaces, while cationic pesticides (bases) will be quite strongly adsorbed on the negatively charged surfaces; like the carboxyl, phenol and hydroxyl functional groups of humic substances (which bear negative charges when the soil pH is above 4) and to clays, oxides and hydroxides, and they may show more correlation with clay content than organic matter content (Wauchope et al., 2002). The effects of acid and base pK values and soil pH on sorption are summarized in Table 2.3.3-1.

Table 2.3.3-1 Mobility of acidic and basic pesticides in soils (Wauchope et al., 2002).



pK_a/pK_b value[a]	*Dominant pesticide species within normal soil pH range (5–8)*	*Environmental consequences*
$pK_a < 3$	X^- (anion)	Highly mobile in soil unless chemical complex formed; less mobile under very acid conditions; very soluble; non-volatile.
$pK_a > 10$	XH (neutral)	Behaves like non-ionic material except under extremely alkaline conditions: less mobile than anion; probably much less soluble than anion; volatility possible.
pK_a 3–10	X^-/XH ratio a *f*(soil pH)	If the value of pH is near pK_a, mobility, solubility and volatility will be sensitive to pH.
$pK_b < 4$	$(XH)^+$ or X^+ (cation)	Quite immobile (clay surface sorption); very soluble; extreme soil sorption leads to long half-life but little biological activity; non-volatile.
pK_b 4–11	$(XH)^+$/X or X^+/X(OH) ratio a *f*(soil pH)	If the value of (14 – soil pH) is near pK_b, mobility, solubility and volatility will be sensitive to pH. Extreme sorption of cation produces an apparent soil 'surface acidity' effect.
$pK_b > 11$	X or X(OH) (neutral)	Behaves like non-ionic material except at extremely acid conditions: much more mobile, and less soluble than cation; volatility possible.

About one in three pesticides are ionisable, i.e. are weakly or strongly acidic or basic; thus partially ionized within the range of normal soil pH. The sorption of such a pesticide will be a combination of the sorption of ionized and unionized species, especially at soil pH close to the pKa-value, where the ratio of ionized and unionized species in solution may change approximately tenfold for each unit change in pH. Whereas strong bases always occur as cations in the normal soil pH-range, and strong acids occur as anions, the ionization of weak bases and weak acids can vary significantly as a function of soil pH (Chaplain et al., 2011). Implications are that sorption (Kd) of ionisable pesticides will correlate poorly with organic matter content if soils with a range of pH values are examined (Wauchope et al., 2002).

Kah et al. (2007) were not able to find a statistically significant effect of sorption on degradation rates for 7 ionisable and biodegradable pesticides in 9 different soils, although there was a strong positive correlation between OC content and degradation rate. The parameters that explained variations in degradation rates depended on the soil-pesticide combinations and had to be evaluated individually. Strong correlation between sorption and degradation of ionisable pesticides are more likely to be found if more similar soil types are compared (Kah et al., 2007) and in soils with small differences regarding pH.

2.3.4 Microbial activity and diversity

Degradation of pesticides in soil is highly dependent on soil microbial populations and activity levels, which in turn are influenced by environmental factors like temperature, humidity and the nutritional status of the soil. Environmental conditions that allow for good growth of microorganisms also favour the biodegradation of pesticides. Functioning of a system is not necessarily coupled with the permanent maintenance of given structural components (i.e. soil microbial community structure), but mixed populations have a greater degradation capacity than individual species due to the diversity of enzyme systems.

Climate has been shown to be a main control of microbial biomass size and activity (Insam, 1990). The general soil microbial characteristics biomass, activity and community structure, have been shown to be linked to more specific environmental detoxification processes like pesticide degradation (Haney et al., 2002; Jones and Ananyeva, 2001; Torstensson and Stenström, 1986). Studies of mineralisation of the herbicide glyphosate in sandy and silt loam soils along a climatic gradient show a good, but site specific, correlation between total organic C-mineralization and pesticide degradation (Figure 2.3.4-1; (Stenrød et al., 2005)). Lab-scale glyphosate mineralization studies in sandy and silt loam soil with comparable characteristics from Grue (Hedmark) and Målselv (Troms) showed consistently higher microbial activities and glyphosate mineralization rates at the southernmost site under a range of winter temperature regimes (Figure 2.3.4-2; (Stenrød et al., 2005)), indicating the effect of other factors than temperature and physio-chemical soil conditions. Preliminary results from a study of the soils from Vollebekk, Særheim, Kvithamar and Holt (cf. Table 2.3.1-1) indicate statistically significant differences in metabolic diversity (i.e. ability to utilize a range of organic C-substrates) between the soils with low and high organic C content (unpublished data). This feature will be studied further to possibly explain the observed differences in pesticide degradation kinetics in these soils (i.e. fenpropimorph, cf. section 2.3.2).

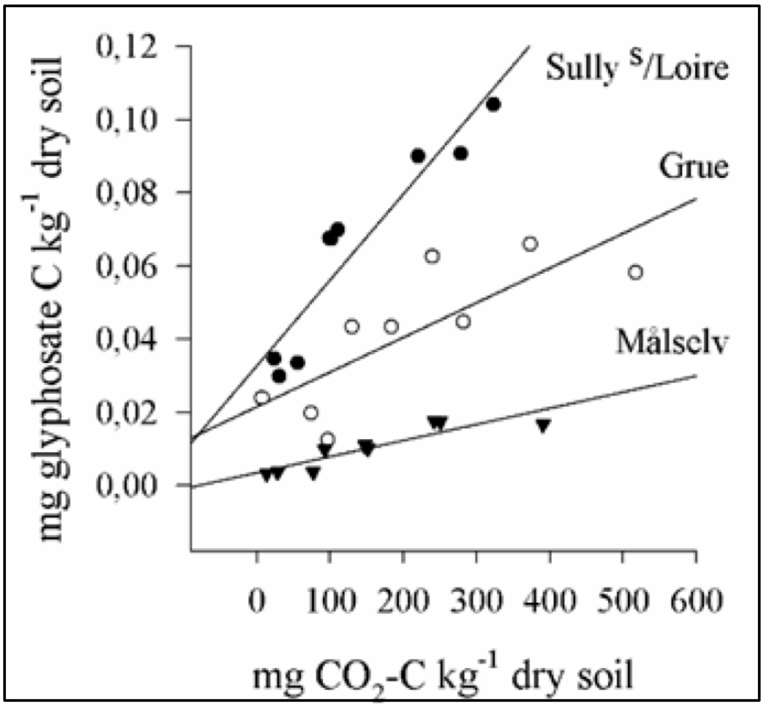


Figure 2.3.4-1 Relationship between total organic C mineralization and total 14C-glyphosate mineralization in soils from Sully s/Loire (FR), Grue (NO, Hedmark) and Målselv (NO, Troms). Total amounts mineralized during a 49 d incubation period.

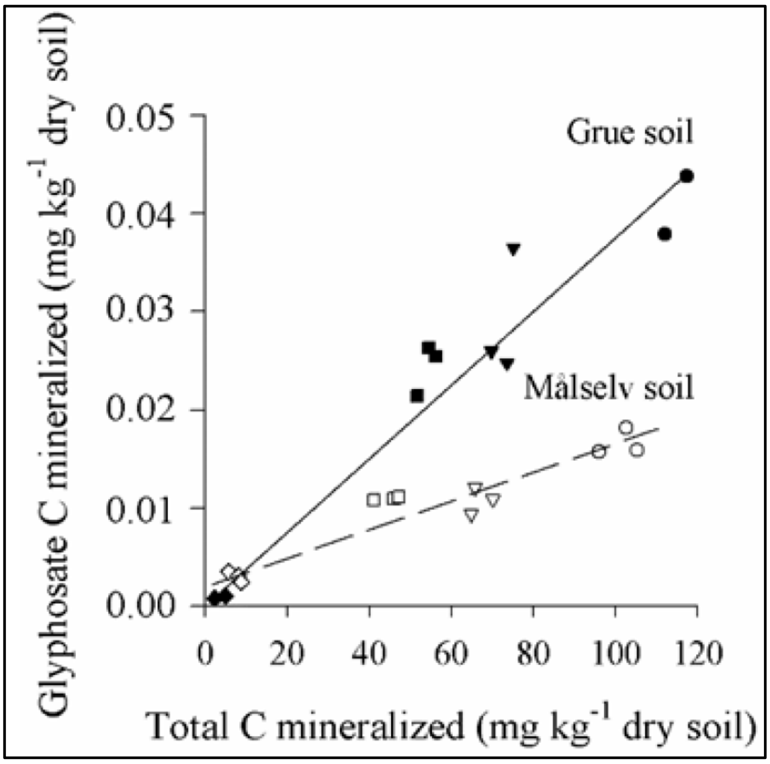


Figure 2.3.4-2 Relationship between total amount mineralized organic C and glyphosate C in Grue and Målselv soils for a range of winter temperature conditions: constant at -5°C (◇ and ◆), 3 wk intervals with incubation temperature of -5°C and +5°C (▫ and ▪), 24 h intervals with incubation temperature of +5 and -5°C (▽ and ▼), and constant at +5°C (o and •). Grue soil is represented by filled symbols and Målselv soil by open symbols. The regression model estimating glyphosate mineralization based on organic C mineralization for the individual soil (r2pred=0.77), is shown separately for Grue (solid line) and Målselv (dashed line) soil.

Variability in soil microbial processes is observed both spatially and over time, and it is found to be higher than the variability in physical and chemical soil properties (Röver and Kaiser, 1999). A study of two Norwegian soils; a silt loam soil in south-east Norway and a sandy loam soil in northern Norway; illustrate the large temporal and spatial variability in some important soil microbial characteristics (Table 2.3.4-1; (Stenrød et al., 2006; Stenrød et al., 2005)). The seasonal and spatial variability in soil microbial characteristics will have a potentially strong effect on the results of pesticide fate studies under different climatic conditions. However, the large inherent variability in soil biological properties might override the effects of climatic or management factors on soil processes.

Table 2.3.4-1 Coefficient of variation (%) of selected soil characteristics assessed through monthly measurements (of bulk soil samples from the plough layer of an agricultural field) through the growing season (May-October) of two consecutive years.

	Soil moisture	Organic C	Microbial biomass	Basal respiration	Diversity index
Seasonal variation					
Grue					
2002	10	45	51	33	26
2003	17	19	19	65	13
Målselv					
2002	31	40	47	16	15
2003	25	29	29	68	11
Spatial variation					
Målselv					
Horisontal (30 cm, small scale)	3		24	81	12
Vertical (30-95 cm, small scale, interrow)	55		145	73	14
Vertical (20-95 cm, small scale, potato row)	62		129	128	19

Studies of a Norwegian silt loam soil showed large variability in glyphosate mineralization rates and total amounts mineralized (3-33 %, coefficient of variation: 37 %) within a single agricultural field in the relatively homogenous upper parts of the plough layer (Stenrød et al., 2006). There are also studies showing the greater spatial variability in pesticide degradation rates in subsoil compared to a mixed topsoil layer (Rodriguez Cruz et al., 2008). This illustrates that results obtained from such laboratory studies with average mixed soil samples should be interpreted with care.

Bioavailability stands out as an important aspect when looking into pesticide persistence in soil, as it affects both its biodegradation and potential eco toxicological effects. Model studies with 2,4-D indicate that localization of microorganisms and pesticides in soil are major driving factors of pesticide biodegradation, by regulating the accessibility to degrading microorganisms and the formation of non-extractable residues (NER) (Pinheiro et al., 2015). Lab studies of glyphosate mineralization in sandy and silt loam soils show a clear relation (positive correlation) between extractability and mineralization rates (Stenrød et al., 2005), with glyphosate being a herbicide that is mainly degraded through co-metabolic processes. Rapid transport of pesticides down the soil profile will increase pesticide persistence due to the low microbial biomass in subsurface soils and, hence, the low exposure of the compound to actively degrading organisms. Similarly, strong sorption of a pesticide to organic or mineral soil constituents will decrease its bioavailability to soil microbes and increase its persistence in soil.

2.4 Climatic factors that affect pesticide degradation rate

2.4.1 Temperature

As DT50 values are more site-specific and heavily influenced by local factors like crop type, soil management, pesticide application time etc., it can be difficult to compare field DT50 values from different sites, unless the fields have been treated in the exact same way as part of carefully planned field dissipation studies. Field degradation studies, in which dissipation is mainly due to degradation, produce DegT50 values that more confidently can be used to compare location effects.

Field degradation studies at four locations from 58°N to 69°N in Norway show that DegT50 of five pesticides that sorb moderately to strongly to soils increase with increasing latitude and with the duration of winter period with cold soils (Almvik et al., in prep.). The main driver for the differences in degradation rates at the Norwegian locations seemed to be the soil temperature. In the laboratory; at a constant soil temperature of 20°C, the pesticide degradation was primarily dependent on the content of organic matter (with a few exceptions), with the slowest degradation in soils with high OC content, probably due to sorption-controlled degradation.

Specific for Norwegian climate is the winter period with near or below-zero soil temperatures when no degradation takes place. As soil temperature decreases from summer until winter, so does soil microbial activity – and with it, the degradation rate of pesticides. Laboratory mineralization studies in Norwegian silt/sandy loam soils have been performed with the herbicides glyphosate (Stenrød et al., 2005) and metribuzin (Benoit et al., 2007; Stenrød et al., 2008), simulating different winter conditions with low temperatures (+5 and -5 °C) and/or alternating freezing and thawing of the soil. These studies show that metribuzin mineralization rates are very slow/negligible at these low temperatures and that alternating freezing and thawing of the soil might induce increased risk of leaching of this mobile substance. Glyphosate mineralization were similarly found to be insignificant during winter, but any effects of freezing and thawing on leaching risk were not distinguishable due to experimental limitations. These studies do however indicate a site specific temperature response in the pesticide degradation processes that may not be sufficiently explained by temperature conditions (cf. section 2.3.4).

Due to the temperature shifts that are typical for the Norwegian climate, one might expect degradation rates to be rapid during summer and slow during winter, leading to a bi-phasic degradation pattern in the field. However, in the Norwegian DegT50 field studies, most of the pesticides showed regular first order degradation kinetics (SFO; Single-First Order). The degradation rates were generally low all year and no rate shifts were observed (Almvik et al., in prep.). Bi-phasic degradation as affected by temporal variation in soil temperature may however be more pronounced for pesticides that do not sorb so strongly to soils as the pesticides in our study.

Temperature normalisation of degradation rates can be performed by using a temperature correction factor, Q10. Q10 is defined as the ratio of pesticide degradation rate coefficients (k2/k1) at a temperature T1 that is 10°C lower than a temperature T2. It is used to normalize degradation rates measured in field studies to a reference temperature (e.g. 20 ^{0}C) and to simulate temperature dependent degradation in environmental exposure modelling. A default Q10 value of 2.2 was used in EU risk assessments until 2007, after which it was re-evaluated and set to 2.58 (EFSA, 2007). High Q10 ratios mean that degradation is highly dependent on soil temperature. As soil temperatures are at or below 10°C for long periods Norway, a correct measure of the Q10 ratio is important. However, Q10 vary from pesticide to pesticide and need to be evaluated for each compound. Studies in four Norwegian soils at 10°C and 20°C, indicated that Q10 values increased from south to north, and values were often much higher than 2.58 (Almvik et al., in prep.).

In conclusion: due to the seasonal temperature variations in fields in Norway, pesticide degradation rates will probably show much more variation in Norwegian sites as compared to European locations. Furthermore, using Q10-values derived from a temperature range of 10-20 ^{0}C to simulate degradation at a much lower range involves an additional uncertainty. The combinations of soils with high OC content and low soil temperatures, as in the northern part of Norway, represent worst case scenarios for degradation of pesticides. The combination of OC-poor soil with low microbial biomass and low to moderate soil temperatures, as in the middle part of Norway, (continental subarctic climate), can lead to very long persistence of pesticides in soils. Soils with high OC content but warm(er) soil temperatures (as in southern Norway with marine west-coat climate, wide-spread also in the central pesticide approval Zone in Europe, see Figure 1-2) may have degradation rates that are comparable to EU data. It would be wise to take the varying climatic zones and agricultural soils in Norway into consideration during the risk assessment before formulated pesticides are approved in Norway. More knowledge on sorption and degradation of pesticides in organic rich soils is necessary.

2.4.2 Precipitation and moisture

With increasing soil moisture, organic matter becomes more hydrophilic and diffusion of hydrophilic pesticides to sorption sites is facilitated (Roy et al., 2000). Hydrophobic pesticides, however, are less sorbed as soil moisture increase because the hydration of the surfaces decreases the accessibility to adsorption sites (Swann and Behrens, 1972). Dry soils favour the sorption of hydrophobic pesticides. Earlier it was believed that microorganisms could only use the dissolved fraction of the compound in soil solution (Cork and Krueger, 1991), but now we know that some degradation of sorbed pesticides cannot be totally excluded (Guo et al., 2000). Soil moisture affects microbial activity and diversity and generally, aerobic microbial activity increases with soil water content up to a maximum point before decreasing (Linn and Doran, 1984).

We are not able to conclude on the importance of precipitation and moisture on degradation in our climate, as there is no data available on the effect of soil moisture on degradation rates of pesticides in Norway.

2.5 Conclusion

What are the most important factors contributing to the degradation of pesticides under Norwegian conditions?

The answer to this question is that the factors that influence degradation rates are specific for each pesticide. To quote: "Soil parameters driving degradation rates depend on the pesticide. The dominance of one process over another depends on the soil-pesticide combination and cannot therefore be generalized" (Kah et al., 2007). Nevertheless, some information can be extracted from Norwegian lab. and field studies: Norwegian laboratory degradation studies indicate that increased soil organic carbon content enhances degradation rates of pesticides that show low sorption (e.g. metalaxyl, bentazone) ,due to increased microbial activity, whereas pesticides that sorb moderately to strongly to soil (e.g. boscalid, propiconazole), display reduced degradation as organic carbon increases as a consequence of sorption and reduced bioavailability. These relationships should be interpreted with care for ionisable pesticides, for which sorption is strongly dependent on soil pH. Some pesticides, especially bases, sorb well to clay minerals, but the effect of clay minerals on degradation vary with soils and must be evaluated on a case-by-case basis.

It should also be remarked that we only have Norwegian laboratory degradation data ($DegT50_{LAB}$) for about 15 out of the 100 pesticides approved for use in Norway, and it is difficult to draw conclusions valid for all pesticides based on a few of them. In most of the Norwegian laboratory degradation studies microbial activity/diversity, aged sorption and formation of non-extractable residues were not studied and the soil organic material was not characterized. This excludes the possibility to specifically analyse the effect of these factors on the degradation

We have even less degradation data from field studies in Norway. Available field degradation data are predominantly based on dissipation studies (DT50) which are difficult to compare between sites or to extrapolate. Some Norwegian field degradation studies ($DegT50_{FIELD}$) have recently been performed, but their interpretation are still in preparation and the results have not been published in journals yet. The preliminary results of the DegT50 field studies show, however, that degradation half-lives observed in laboratory studies are not necessarily comparable to field situations – which further complicates an assessment of factors important for the degradation of pesticides under Norwegian conditions.

The recent DegT50 field studies display a large variation in fungicide degradation rates from Klepp in the south to Tromsø in the north. The climate (temperature) seems to be more decisive for fungicide degradation than the soil type. The climate (temperature) seems to be more decisive for fungicide degradation rates than the soil type. Temperature probably affects degradation rates both directly, by affecting microbial metabolic processes, and indirectly, by influencing long-term soil formation processes with consequences e.g. for the properties of the organic matter. Normalisation of DT50-values using a Q10 derived from laboratory studies will only account for the direct effect of temperature. The understanding of climate effects on microbial activity and diversity as well as degradation capacity is far from complete.

Fungicide degradation was slow at two sites having low soil temperatures, even though microbial biomass was hugely different at the sites. This may be due to difference in microbial activity, soil type or crops.

Targeted risk assessment

Ideally, for risk assessment of pesticides in Norway, the varying climatic zones as well as the diversity in agricultural soils and crops should be taken into consideration when degradation rates are estimated for use in pesticide fate modelling. In addition, vulnerable areas with slow degradation and/or high leaching/runoff risk should be recognized. A database with representative soils and climates for various crops should be established in Norway and utilized in a targeted risk assessment approach. Then, the degradation of pesticides to be used in for example fruit/berry cropping, could be evaluated in respect to representative and vulnerable soils and climates in fruit/berry regions in Norway.

3 Updating the existing dataset on the range of soil and climate parameters in Norway

New updates of soil and climate for the most important agricultural regions in Norway are given in the following chapter (Table 3.1-1). Dominating soil types for each region are described together with the accompanying climate largely contributing to the development of the soil profiles. New updates are especially given for the South West and Northern parts of Norway with organic rich soil types (Umbrisols and Histosols).

3.1 Ranges in soil parameters in Norwegian agriculture

NIBIO (former Norwegian Institute of Forest and Landscape) has been mandated to implement national mapping and monitoring program and adapted the international soil classification system World Reference Base of Soil Resources (WRB) as a national system for classifying soils.

Until 2008 (Sperstad and Nyborg, 2008), 13 WRB groups and 270 soil units have been mapped on agricultural lands in Norway. This report gave an overview of these groups and the 50 most important soil units, with descriptions of soil characteristics and properties in relation to agricultural use as well as their occurrence and distribution.

Table 3.1-1 Selected WBR units in Norwegian agricultural land compared with Europe and globally (Solbakken et al., 2006).

WBR group	% of agricultural area	% land area	% land area
	Norway*	Europe**	World***
Albeluvisol	21.6	14	2
Stagnosol	22.4	No information	No information
Cambisol	21.5	12	12
Leveled soil	9.4	No information	No information
Arenosol	5.5	1	7
Gleysol	5.1	5	6
Umbrisol	2.1	3	1
Podzol	2.3	14	4
Histosol	2.6	5	3
Fluvisol	1.1	5	3
Leptosol	0.5	9	13

*Source: NIJOS

**Source: European commission, 2008. “Soil Atlas of Europe”

*** Source: FAO, 2001. “Lecture notes on the major soils of the world”

NIJOS / Norwegian Institute of Forests and Landscape (later Nibio) has at the end of 2015 surveyed more than half of Norway's farmland. The smallest polygon of the mapping unit is 4 decare and nearly 2,000 mapping units (soils) are registered. Soil types are documented with a representative data set, which includes designations and horizon thicknesses for each layer, the grain size distribution, organic carbon, and for a major part of soil types, exchangeable cations.

3.1.1 Properties of the top 3 soil dominating soil types in Norway

Two of the most dominant agricultural soils in Norway are described as Nordic Reference Soils (Greve et al., 1998). Reference soil no 10 (Ås) is a clay loam belonging to Albeluvisols (Figure 3.1.1-1), and reference soil no 8 (Hole) is a sandy soil belonging to Cambisols.

Figure 3.1.1-1 Soil profile representing an albeluvisol (Solbakken et al., 2006) (Photo: Eivind Solbakken).

Organic Carbon in the Ap-layer is 2.5-2.6 %. In the other soil layers the content of org. C usually will be from 0.1-0.5%. pH differs from 6-7 in the total soil profile. This soil type covers 36 % of the agricultural area in Østfold (Sperstad and Nyborg, 2008) and 28 % of the agricultural area in Vestfold (Solbakken et al., 2006). The most frequently subunit of Albeluvisol, Epistagnic Albeluvisol (Siltic) can be found north to Trøndelag and Troms. 700 km^2 of this soil type was mapped in 2007 (Sperstad and Nyborg, 2008). This soil type is also represented in the National scenario of Norway for prediction of groundwater concentration (Rustad) simulated with the MACRO model (Eklo et al., 2008). In addition this soil type is covered by the Syverud scenario which is a drainage scenario with WISPE.

Figure 3.1.1-2 Areas with Fluvic Cambisols along Glomma (Photo E. Solbakken).

Cambisols (Figure 3.1.1-2) represented with reference soil 8 from Hole have organic in the Ap-layer from 1.5-2.5%. In the other soil layers the content of org. C usually will be from 0.1-0.5%, but this is sediments from rivers so it can occur that in deepen layers the content of org. C can be about 1%. pH: 5-6 in the total soil profile. The WBR unit is Fluvic Cambisol or Endostagnic Fluvisol. 130 km^2 has been mapped of this soil type and are important areas for potato growing along the big rivers (Figure 3.1.1-1). This soil type has been used in several field studies in Norway (Stenrød et al., 2007) and represents vulnerable areas with high risk of leaching to groundwater.

The third most important soil type in Norway is Stagnosols (Figure 3.1.1-3), a group containing soil with poor drainage properties because of the occurrence of soil horizons of low permeability preventing leaching. This soil type is distributed most frequent on outside of the moraine ridges (Raet) and cover almost 30 % of the agricultural area in Østfold and Vestfold. 585 km2 has been mapped of the soil classified as Luvic Stagnosol. This soil type is also represented in the National scenario for leaching to groundwater (Heia scenario) (Eklo et al., 2008). This soil has 2 % organic carbon in the plow layer. The top layer is dominated by a sandy and silty soil, with increasing clay content with depth.

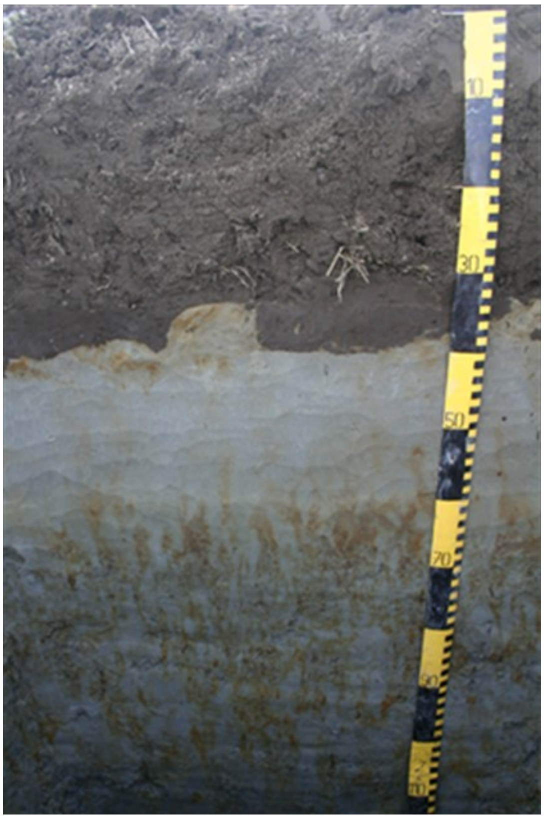


Figure 3.1.1-3 Typical soil profile of the WRB group Stagnosol (Photo Siri Svendgård-Stokke).

Some other important soil types are Romeriksmjæle (silty soils) which occurs in an important agricultural area Romerike between Oslo and the lake Mjøsa, classified as Planosols containing 2.5 % organic carbon in the Ap-layer In the other soil layers the content of org. C usually will be from 0.1-0.5% ,pH: ca 6 in the total soil profile. Umbrisols and Histosols with high content of organic matter are important for South west Norway and the coastline of North Norway and described in the end of the chapter 3.1.3.

There are mostly glacial deposits (moraines) in the areas around the lake Mjøsa (Lillehammer, Østerdalen, Gausdal). The deposits can be either oligotrophic or nutrient-rich: Nutrient-rich: Phaeozems Organic Carbon: Ap-layer: 2.0-3.5 %. In the other soil layers the content of org. C usually will be from 0.1-0.5%; pH: 6-7 in the total soil profile. Oligotrophic: Dystric Cambisol. Organic Carbon: Ap-layer: ca 2.0 %. In the other soil layers the content of org. C usually will be from 0.1-0.5%; pH: 5-6 in the total soil profile.

3.1.2 Properties of the most important Norwegian soils

Organic carbon in the Ap layer is often from 1.5-3 %, seldom above 4 %. In the other soil layers the contents of organic C usually will be between 0.1 and 0.5%. Exceptions are the organic soil, Umbrisols and Histosols. pH is generally between 6 and 7 in the whole soil profile. In oligotrophic moraine and in sandy soils the pH can be as low as 5. Only in typically marine sediments with much sand from shell etc. pH is higher (up to 8 seldom). CEC (cation exchange capacity) is not investigated in the same way as pH and organic matters. CEC is only investigated in some few selected soil profiles in Ås, Hole og Heiabekken. CEC increases usually with increasing contents of clay and organic matter. Generally CEC varies from 7-9 in the Ap-layer and CEC < 4 mekv/100g in layers below Ap. In clay soils it is otherwise. Here CEC is13 in the Ap – layer and CEC >7 in all the layers below.

3.1.3 Distribution of WRB groups in selected Norwegian regions

Distribution of soil types in Norway differs largely between the different regions, which is related to different origin, geology and soil generating processes after the ice cover 10 000 years ago. These differences become more visible when looking at the frequency and area of the different soil types splitting the country into five main regions: Eastern Norway south, Eastern Norway north, Rogaland, Trøndelag, North of Norway (Tables 3.1.3-1 to 3.1.3-5). The distribution of the soil types in the different regions are based on areas mapped pr. January 1, 2014 and classification after WBR 2006.

Eastern Norway south (Table 3.1.3-1) consists of Østfold, Vestfold, Akershus, Oslo, Buskerud without Hallingdal and eastern and southern parts of Telemark. The total area mapped is 2,577 km^2.The dominating soil type inside the moraine ridge (raet) in Østfold and Vestfold is Albeluvisol with origin of marine clay and deposits. These areas represent the oldest and "mature" agricultural landscape in Norway. Globally Albeluvisols is prevalent in the boreal zone and in the coldest part of the temperate zone.

Table 3.1.3-1 Mapped area of agricultural land in Eastern Norway (south) pr. 1.1.2014

WRB-gruppe	Area (km^2)	%
Albeluvisol	813,0	31,6
Stagnosol	696,2	27,0
Levelled soil	348,1	13,5
Cambisol	292,2	11,3
Arenosol	119,2	4,6
Gleysol	108,1	4,2
Podzol	58,1	2,3
Umbrisol	38,0	1,5
Histosol	32,6	1,3
Fluvisol	30,1	1,2
Regosol	13,0	0,5
Phaeozem	12,8	0,5
Anthrosol	8,5	0,3
Leptosol	7,0	0,3

Eastern Norway north consists of Hedmark and Oppland. Areas mapped are 1123 km^2. On cultivated land in the inner parts of Eastern Norway, Cambisol is the dominant group (Table 3.1.3-2). Cambisol is one of the most prevalent WRB groups worldwide. The prevalence is largest in the boreal and temperate zones and especially in the areas affected by last glaciation. In Europe Cambisols cover approximately 12 % of the land area. (Solbakken et al. 2006).

Table 3.1.3-2 Mapped agricultural land area in Eastern Norway (north) pr. 1.1.2014.

WRB-gruppe	**Area (km^2)**	**%**
Cambisol	602,5	53,6
Phaeozem	134,2	12,0
Stagnosol	129,7	11,6
Arenosol	56,4	5,0
Histosol	35,7	3,2
Regosol	32,5	2,9
Albeluvisol	22,1	2,0
Gleysol	21,7	1,9
Podzol	20,9	1,9
Fluvisol	19,7	1,8
Levelled soil	18,8	1,7
Anthrosol	16,0	1,4
Umbrisol	7,0	0,6
Leptosol	6,2	0,6

Rogaland: In Rogaland most of Jæren is mapped, totally 256 km^2 (Table 3.1.3-3). Umbrisol is the most prevalent soil in the southwest of Norway (Rogaland). In Norway we find most Umbrisols close to the coast, particularly in the wet areas and areas with steady inflow of fresh water. Globally Umbrisols are rare, but appear on the west coast of Portugal and Spain. In its natural state Umbrisols have low pH due to nutrient-poor parent material and vegetation that emit acidic organic material. Umbrisols are often formed under cool and humid climates from glacial tills. Degradation of organic material occurs slowly and organic matter in the plow-layer is often more than 10%.The nutrients are released and quickly washed out (Solbakken et al., 2006).

Table 3.1.3-3 Mapped agricultural land area in Rogaland pr. 1.1.2014

WRB-gruppe	Area (km^2)	%
Cambisol	602,5	53,6
Phaeozem	134,2	12,0
Stagnosol	129,7	11,6
Arenosol	56,4	5,0
Histosol	35,7	3,2
Regosol	32,5	2,9
Albeluvisol	22,1	2,0
Gleysol	21,7	1,9
Podzol	20,9	1,9
Fluvisol	19,7	1,8
Levelled soil	18,8	1,7
Anthrosol	16,0	1,4
Umbrisol	7,0	0,6
Leptosol	6,2	0,6

Trøndelag. The mapped area in Trøndelag comprises parts of Namdalen and Innherred in Nord Trøndelag, and Trondheim, Melhus, Klæbu, Malvik, Skaun, Ørlandet and Oppdal in Sør Trøndelag. Totally 842 km^2 is mapped (Table 3.1.3-4). The most prevalent soil in this area is Stagnosols. The origin of Stagnosols is usually related to young materials in a flat to gently sloping landscape. In the northern hemisphere, they appear in the boreal belt, right up to the tundra areas. Information of the distribution of this soil type globally is limited as this soil type is new in the classification system (Solbakken et al., 2006). Characteristic for areas with this soil type is that surface water accumulates in the cracks and pores down to 50 cm depth in periods after rainfall or snowmelt. The reason might be buried layers with low hydraulic conductivity down into the profile retarding the water movement.

Table 3.1.3-4 Mapped agricultural land area in Trøndelag pr. 1.1.2014.

WRB-gruppe	Area (km^2)	%
Stagnosol	239,7	28,5
Cambisol	143,6	17,1
Gleysol	74,9	8,9
Albeluvisol	73,5	8,7
Levelled soil	69,6	8,3
Arenosol	58,5	7,0
Regosol	39,4	4,7
Histosol	39,1	4,6
Anthrosol	39,0	4,6
Umbrisol	33,0	3,9
Phaeozem	10,4	1,2
Leptosol	9,9	1,2
Fluvisol	6,9	0,8
Podzol	4,5	0,5

North Norway. The mapped area in this region comprises Lofoten, Ofoten, parts of Tromsø and some scattered areas in Nordland, Troms and Finnmark. Total mapped area is 111 km^2 (Table 3.1.3-5). The most prevalent soil is Histosol, which in national scale represent less than 2 % of the agricultural area. Globally this soil type cover less than 3 % of the land area and most of the areas are in the subarctic region. Characteristic for Histosol is the organic layer of >40 cm, sometimes buried under a thin layer of mineral soil. Histosol is formed when contribution of organic matter is larger than the microorganisms in soil are able to degrade.

Table 3.1.3-5 Mapped agricultural land area in North Norway pr. 1.1.2014. (This table does not represent the entire area of North Norway as half of the mapped area comes from Lofoten where also *potential* agricultural soil is mapped and included).

WRB-gruppe	Area (km^2)	%
Stagnosol	239,7	28,5
Cambisol	143,6	17,1
Gleysol	74,9	8,9

WRB-gruppe	Area (km^2)	%
Albeluvisol	73,5	8,7
Levelled soil	69,6	8,3
Arenosol	58,5	7,0
Regosol	39,4	4,7
Histosol	39,1	4,6
Anthrosol	39,0	4,6
Umbrisol	33,0	3,9
Phaeozem	10,4	1,2
Leptosol	9,9	1,2
Fluvisol	6,9	0,8
Podzol	4,5	0,5

3.1.4 Organic agricultural soils

In South Eastern Norway more than 90 % of the mapped agricultural area has less than 6 % organic matter. Areas with organic soil increase along the west coast towards the north with exceptions for Trøndelag, which is more like Eastern Norway (Table 3.1.4-1). Humic rich soil covers more than 60 % of the agricultural areas in Rogaland

Table 3.1.4-1 Classes of soil types with organic matter in the plow-layer in different regions of Norway (% of mapped area). (This table does not represent the entire area of North Norway as half of the mapped area comes from Lofoten where also *potential* agricultural soil is mapped and included).

Region	Organic soils (> 20 % OM)	Humic rich soils (6 -20 % OM)	Others (< 6 % OM)
Eastern Norway south	1,5	4,9	93,6
Eastern Norway north	3,4	3,7	92,8
Rogaland	17,5	61,0	21,6
Trøndelag	6,0	9,1	84,9
Nord Norge*	40,3	14,6	45,1

3.1.5 Distribution of the most common soil types

Summary: Each agricultural region in Norway is dominated by one specific soiltype for each region. Albeluvisol, Cambisol, Umbrisol, Stagnosol and Histosol in respectively Eastern Norway south, Eastern Norway north, Rogaland, Trøndelag and North of Norway. New updates for Norway include especially Umbrisols and Histosols rich in organic matter. Albeluvisols, Cambisols and Stagnosols are representing the main soil types in the agricultural area in Norway. These are also included in the groundwater (Rustad and Heia) and surface water scenarios (Syverud) developed for Norway. Experience from pesticide fate in the organic rich soils on the south west coast and north of Norway is limited.

3.1.6 Crops

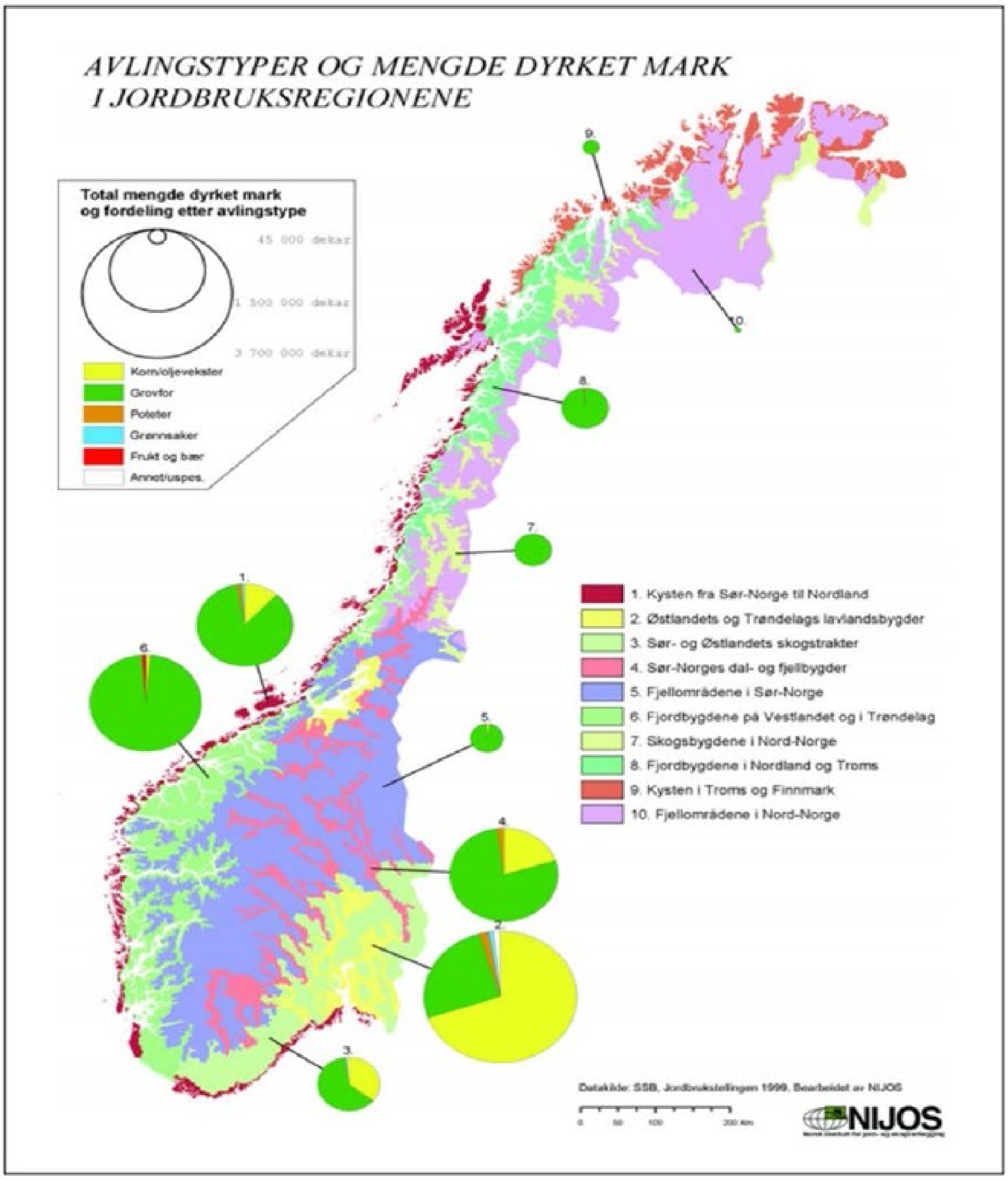


Figure 3.1.5-1 Distribution of crops in the different regions of Norway.

Some statistics are available describing the distribution of different crops to regions (Figure 3.1.5-1), but the connections to soil types are not coupled yet. The distribution and type of production might be based on soil properties, but can be politically explained by use of incentives in different regions. Combining soil types and statistics on agricultural productions could bring important information of constraints of pesticide use in vulnerable regions especially in region with crops (vegetables, fruit and berries) with frequent use of pesticides.

3.1.7 Norwegian soil types and soil types used in EU scenarios

Table 3.1.6-1 Topsoil primary properties of selected European, Nordic and Norwegian soil.

Source	Representative field site	Organic carbon %	Texture class	Clay[1] %	Silt[1] %	Sand[1] %	pH	Bulk density $g\ cm^{-3}$
Focus SW Scenario[2]								
D1	Lanna	2.0	Silty clay	47	46	7	7.2	1.35
D2	Brimstone	3.3	Clay	54	39	7	7.0	1.20
D3	Vredepeel	2.3	Sand	3	6	91	5.3	1.35
D4	Skousbo	1.4	Loam	12	37	51	6.9	1.48
D5	La Jailliere	2.1	Loam	19	39	42	6.5	1.55
D6	Váyia, Thiva	1.2	Clay loam	30	34	36	7.5	1.43
R1	Weiherbach	1.2	Silt loam	13	82	5	7.3	1.35
R2	Valadares, Porto	4.0	Sandy loam	14	19	67	4.5	1.15
R3	Ozzano, Bologna	1.0	Clay loam	34	43	23	7.9	1.46
R4	Roujan	0.6	Sandy clay loam	25	22	53	8.4	1.52
Norwegian SW Scenarios[3]								
	Bjørnebekk	1.5	Silty clay loam	26	64	9	6.0	
	Syverud	3.1	Loam/silt loam	27	47	26	5.5	
FOCUS GW scenarios[4]								
	Châteaudun	1.4	Silty clay loam	30	67	3	8.0	1.3
	Hamburg	1.5	Sandy loam	7	25	68	6.4	1.5
	Jokioinen	4.1	Loamy fine sand	4	23	73	6.2	1.3
	Kremsmünster	2.1	Loam/silt loam	14	50	36	7.7	1.4
	Okehampton	2.2	loam	18	43	39	5.8	1.3
	Piacenza	1.0	loam	15	45	40	7	1.3
	Porto	3.8	loam	10	48	42	4.9	0.9
	Sevilla	0.9	Silt loam	14	51	35	7.3	1.2
	Thiva	0.7	loam	25	43	32	7.7	1.4
Norwegian GW scenarios[3]								
	Rustad	1.9	Silty clay loam	13	60	27	6.6	1.3
	Heia	2.2	Sandy loam	5	30	65	6.4	1.4
Nordic Reference soils[5]					(0.002-0.06 mm)	(0.06-2mm)		
1a	Jyndevad	1.4	Loamy sand	5	5	90	6.1	
2a	Flakkebjerg	1.1	Loam	14	34	52	6.4	
3a	Borris	1.5	Sandy loam	8	20	72	6.8	
4a	Mikkeli	2.8	Sandy loam	5	32	63	6.1	
6a	Ylistaro	4.3	Silt loam	24	64	12	4.3	
8a	Hole	2.0	Sandy loam	6	42	52	6.7	
10a	Ås	2.4	Loam	23	40	37	5.9	
13a	Lanna	2.1	Silty clay	47	45	8	6.5	
Norwegian field trial sites								
	Vollebekk	1.6	Loam	25	46	30		
	Særheim	3.3	Sandy loam	11	32	57		

Source	Representative field site	Organic carbon %	Texture class	Clay[1] %	Silt[1] %	Sand[1] %	pH	Bulk density $g\ cm^{-3}$
	Kvithamar	1.8	Sandy loam	7	49	45		
	Holt	5.5	Sandy loam	7	35	58		
	Kroer	2.5	Loam	19	45	36	5.5	
	Skuterud	1.9	Silty clay loam	27	60	13	5.8	
	Målselv	0.9	Sandy loam	3	34	64	6.6	
	Grue	0.9	Silt loam	4	49	47	5.9	
	Askim	1.0	Clay loam	24	53	24	6.7	
	Rygge	1.1	Sandy loam	9	27	65	6.5	
	Roverud	1.4	Silt loam	10	73	17	6.5	

3.2 Ranges in climatic parameters in Norwegian agriculture

Table 3.2-1 Temperature data, normal period (1961-1990) and more recent data (1991-2014), for selected sites within important agricultural regions in Norway.

Site	**Period**	**Mean daily temperature (°C)**			
		Annual	Mar-May	Jun-Sep	Oct-Feb
Holt (North)	Normal 1961-1990	3.1	1.2	9.8	-1.2
	1991-2014	3.8	2	10.3	-0.1
Kvithamar (Central)	Normal 1961-1990	5.0	4.3	12.3	-0.3
	1991-2014	5.9	5	13.2	0.8
Særheim (West)	Normal 1961-1990	7.1	5.7	13.0	3.2
	1991-2014	8.1	6.5	13.7	4.0
Apelsvoll (East, northern)	Normal 1961-1990	3.6	2.9	12.8	-3.3
	1991-2014	5.3	4.3	13.6	-1.9
Landvik (East, southern)	Normal 1961-1990	6.9	5.5	14.5	1.9
	1991-2014	7.7	6.6	15.1	2.4

Table 3.2-2 Precipitation data, normal period (1961-1990) and more recent data (1991-2014), for selected sites within important agricultural regions in Norway.

Site	**Period**	**Mean summed precipitation (mm)**			
		Annual	Mar-May	Jun-Sep	Oct-Feb
Holt (North)	Normal 1961-1990	1000	175	310	515
	1991-2014	953	192	291	470
Kvithamar (Central)	Normal 1961-1990	900	158	363	379
	1991-2014	970	197	352	421
Særheim (West)	Normal 1961-1990	1280	210	455	615
	1991-2014	1405	230	456	719
Apelsvoll (East, northern)	Normal 1961-1990	600	105	275	220
	1991-2014	699	125	287	287
Landvik (East, southern)	Normal 1961-1990	1230	225	412	593
	1991-2014	1359	229	429	701

Comparing the “normal” temperature and precipitation from 1961 to 1990 with a “new normal” from 1991 to 2014, the climate has changed (Tables 3.2-1 and 3.2-2). For the previous described agricultural areas in Norway, annual temperature has increased for all five regions and seasons (Table 3.2-1). The rainfall has increased for all seasons and regions except for the Northern Norway (Holt in Tromsø) and summer season at Kvithamar (Trøndelag) with lower precipitation in June to September.

4 Use of normalized field data from Norway and Europe

4.1 Are normalized field data from abroad acceptable for use in Norway?

DegT50 values for pesticides in soils are important input values in the current FOCUS modelling frameworks for estimating surface water and groundwater exposure levels (FOCUS, 2001; FOCUS, 2014) and in the soil exposure scenarios developed by EFSA (EFSA, 2015). The DegT50 values used as input parameters can either be normalized field DegT50 values or laboratory DegT50 values.

In field conditions, soil temperature and moisture are highly variable with seasons and this affects the pesticide degrading microorganisms accordingly. Through a "time-step normalization" approach, field sampling day lengths are adjusted into normalized day lengths, representing day lengths at the reference conditions; soil temperature of 20°C and soil moisture at pF = 2. The conversion (or "inverse modelling") of field soil temperature and moisture data into reference conditions are performed by the use of the Arrhenius and Walker equations, respectively. The corrected day lengths are used to determine the cumulative time between sampling points and the latter used as input into a standard kinetic evaluation using ModelMaker to produce new degradation curves and DegT50 values.

DegT50 values for five pesticides in four fields in Norway have been time-step normalized to reference conditions (Almvik et al., in prep.)

The normalization of field data seemed to harmonize DegT50 values from the various sites - which was expected, as soil temperature and moisture impacts have been normalized to reference conditions. Yet, some contrasts are still seen among the DegT50 values at the sites, especially for the fungicides propiconazole, boscalid and pyraclostrobin. These discrepancies may either be due to a) soil-specific characteristics or b) the normalization procedure not being fit for adjusting cold climatic areas with freezing soils (see part 4.2). The normalized DegT50 in Norwegian soils showed some agreement with normalized EU endpoints, at least for those pesticides that are known to degrade moderately fast (bentazone and fenpropimorph). However, the numbers of EU endpoints are few – and (or) not readily available for the research community. The EFSA guidance document for evaluating laboratory and field dissipation studies to obtain pesticide DegT50 was launched in 2014, and we believe normalized EU field DegT50 values will be more abundant within a few years. Our Norwegian DegT50 values for propiconazole seem to be the first values of its kind, as no EU endpoints exist. There is also a need to produce more Norwegian DegT50 values in order to be able to make decisive conclusions on pesticide degradation rates in Norwegian climate. The degradation rates of boscalid in the Stjørdal and Tromsø soils were found to be very slow and did not match the EU endpoints. Actually, BASF found similar zero degradation of boscalid in a field in Skåne in 1998 (Bayer and Grote, 2001), but these results were never included in the EU assessment of the compound. . The results show that some pesticides can be much more persistent in our Northern Zone climate than in southern parts of EU, and calls for more information about the degradation of pesticides in Norway and the Northern Zone.

4.2 Assumptions and simplifications in the time-step normalization procedure

"In general, DegT50matrix values from field studies are expected to be lower than DegT50matrix values from laboratory studies, but the opposite may happen occasionally" (EFSA, 2014).

The field DegT50 values determined in our studies (Almvik et al., in prep.) were however overall very much shorter than the corresponding laboratory values. For a mobile pesticide like bentazone this is not surprising, as rapid leaching of bentazone took place in the fields, but not in the laboratory incubation studies. But leaching cannot explain the short DegT50field for the fungicides, as they were detected predominantly in the top 0-30 cm layer, with some leaching down to 50 cm. Hence, assumptions and simplifications in the time-step normalization procedure need to be examined:

- A moisture-dependency parameter set to 0.7 (assumed default) was applied in the Walker equation for normalizing soil moisture.
- The Arrhenius equation applies only to moderate soil temperatures (generally 10 to 50°C) where soil enzymes are stable (Paul and Clark, 1998; Tabatabai, 1994). In our field studies, soil temperatures were usually down or below zero during winter.
- There is a lack of knowledge on the effect of temperature on degradation in the temperature range 0-10°C.
- Q10 values determined in the laboratory degradation studies (at 20 and 10°C), if available, were used as input in the Arrhenius equation for normalizing soil temperature. For those pesticides where Q10 values were not available, a default Q10 of 2.58 was applied. The laboratory Q10 values were mainly larger than 2.58, resulting in significantly shorter day lengths and higher degradation rates than if the default values had been used.
- Normalization of soil temperature and moisture should be performed "within the layer of soil containing the bulk of the pesticide residues" (FOCUS, 2006). The topsoil (0-10 cm) was found to contain the bulk (approx. 70%) of the fungicide residues, hence soil temperature and soil moisture for this layer were used in this normalization. The newer EFSA guidance relates DegT50 to the 0-30 cm soil layer, thus it could be considered to use soil temperature and moisture at 0-30 cm depth for all sites.
- Several field DegT50 values were considerably shorter than the lab DegT50 values, i.e. even before normalization. Lab and field degradation data are not necessarily

comparable, as some soil factors are more pronounced in laboratory degradation studies than in field degradation studies.

- The EFSA Guideline (EFSA, 2014) has a range of options to be tested in order to verify or reject field-derived DegT50matrix for the benefit of lab.-derived DegT50matrix, but these procedures require re-modelling degradation kinetics with ModelMaker and were not tested.

Conclusion: the fact that the normalized DegT50 values are a lot shorter than the laboratory values at the same reference conditions may point to some systematic error in the normalization procedure (e.g. the default simplifications in the Walker and Arrhenius equations), or that the parameters affecting degradation in the laboratory are different from the parameters that affect degradation in the field. Consequently, lab-derived and field-derived DegT50matrix values should be compared and interpreted with care.

The large variations in normalized DegT50 values obtained in field studies in Norway as well as in other regions in Norway cannot be explained by differences in the associated parameters characterizing the soil and microbial community. It is therefore not possible to determine if a certain field study is more or less representative for "Norwegian conditions". As a conservative approach, the highest, normalized DegT50 from the European field studies should be selected for the Norwegian risk assessment independent on geographic vicinity. As an alternative, when a sufficient number of data are available, a high percentile (e.g 80 or 90-percentile) should be used rather than the geomean.

5 Parameters used in modelling scenarios for exposure assessment (soil, climate and agronomic)

5.1 Guidance for parametrization

The first guidance document for exposure assessments was developed by "The Forum for the Co-ordination of pesticide fate models and their Use" (FOCUS, 1997). The risk assessment is following the principles of tired approach and the general principles of tiered exposure approaches are: 1. Lower tiers are more conservative than higher tiers, 2.Higher tiers are more realistic than lower tiers, 3. Lower tiers usually require less effort than higher tiers, 4. In each tier all available relevant scientific information is used and 5. All tiers aim to assess the same exposure goal. Conservative is defined in the context of this opinion as 'on the safe side with respect to the risk assessment' (EFSA, 2010b). EFSA (EFSA, 2010a) developed this scenario selection procedure.

The values from the laboratory and field experiments are currently used in risk assessment of plant protection products. This includes the FOCUS modelling framework with different models for estimating surface water and groundwater exposure levels (FOCUS, 2001; FOCUS, 2009). In addition they are used in the revised Guidance document on Persistence of Pesticides in Soil (EFSA, 2015). The aim of part of the EFSA report from 2014 (EFSA, 2014) is to provide guidance on selecting the appropriate input values for exposure modelling.

For exposure assessment in soil, the half-life at 20 degrees in topsoil at field capacity is used as input parameters for simple and numerical models. If the half-life of lab studies exceed 60 days, four or more field dissipation studies are required. If there are persistent compounds with longer than one year for 90 % dissipation in the field additional one or two soil accumulation studies is needed. For all relevant tiers a stepped approach is proposed for estimation of DegT50. Step 1 considering values from lab studies. Step 2 DegT50field including field dissipation studies. Step 3 including additional values from accumulation studies. The reason for that is that field dissipation studies and soil accumulation studies provide more realistic estimates for half-life than laboratory studies. These endpoints are required for use in risk assessments of soil exposure assessment, but also for the assessment of leaching to groundwater and surface water.

5.2 Prediction of concentration in soil

According to the work sharing document for the Northern zone (GUIDANCE DOCUMENT, 2014), if representative field data are available, the worst case DT50 field (non-normalized) should be applied. If no representative field data are available a worst case DT50lab (normalized) should be used. If field studies are used, it must be scientifically justified that these are representative to Northern zone member state conditions with regard to soil, type, pH and climate). For the northern zone the Finnish PEC soil calculator should be used (http://www.tukes.fi/pecsoilcalculator). The worst case DT50 value from laboratory should primarily be used as an input value. If DT50 field values exist, normalized values can be used case by case. The Finnish PECsoil-calculator considers only Single First Order (SFO) degradation kinetics. Calculation of the baseline plateau PECsoil for the last years have used the non-normalized worst case DT50field or worst case DT50lab normalized to 10˚C and a soil depth 5 cm. The different Nordic countries have national cut-off criteria especially for non-professional use.

5.3 Leaching to groundwater

Nine realistic worst case groundwater scenarios (FOCUS scenario) have been defined to represent agriculture in EU (Figure 5.3-1). In the Northern zone two types of scenarios are used in the process of pesticides registration within the Northern zone. Two FOCUS groundwater modelling scenarios are used as a first step in the assessment of potential risk to groundwater in registration of pesticides. These two scenarios are Hamburg and Jokioinen, which are modelled with PEARL and PELMO. Soil properties and weather data have been defined for the FOCUS scenarios.

Location	Mean Annual Temp. (°C)	Annual Rainfall (mm)	Topsoil	Organic Matter (%)
Châteaudun	11.3	648	silty clay loam	2.4
Hamburg	9.0	786	sandy loam	2.6
Jokioinen	4.1	638	loamy sand	7.0
Kremsmünster	8.6	900	loam/silt loam	3.6
Okehampton	10.2	1038	Loam	3.8
Piacenza	13.2	857	Loam	1.7
Porto	14.8	1150	Loam	6.6
Sevilla	17.9	493	silt loam	1.6
Thiva	16.2	500	Loam	1.3

Figure 5.3-1 Location of the 9 groundwater scenarios in EU (FOCUS, 2000).

The second step is using national scenarios for the exposure calculations. Some members of the Northern Zone also have specific national modelling scenarios. Sweden (Krusenberg, Näsbygard, Önnestad), Norway (Heia and Rustad) and Denmark (Karup and Langvad). Input files corresponding to the nine FOCUS scenarios have been developed for use with the simulation models PEARL, PELMO and PRZM, whilst input files for the national scenario have been developed for the model MACRO. For Norway, two scenarios have been selected and established, Rustad and Heia with use of MACRO (Eklo et al., 2008). The different soil types are put together in the soil texture triangle and show that Rustad is close to Chateaudun (Figure 5.3-2). Heia has the same soil texture as Jokioinen and Hamburg.

Table 5.3-1 Topsoil primary properties of soils in groundwater scenarios in Europe and Norway.

Source	Representative field site	Organic carbon %	Texture class	Clay [1] %	Silt [1] %	Sand [1] %	pH	Bulk density g cm^{-3}
FOCUS GW scenarios[4]								
	Châteaudun	1.4	Silty clay loam	30	67	3	8.0	1.3
	Hamburg	1.5	Sandy loam	7	25	68	6.4	1.5
	Jokioinen	4.1	Loamy fine sand	4	23	73	6.2	1.3
	Kremsmünster	2.1	Loam/silt loam	14	50	36	7.7	1.4
	Okehampton	2.2	loam	18	43	39	5.8	1.3
	Piacenza	1.0	loam	15	45	40	7	1.3
	Porto	3.8	loam	10	48	42	4.9	0.9
	Sevilla	0.9	Silt loam	14	51	35	7.3	1.2
	Thiva	0.7	loam	25	43	32	7.7	1.4
Norwegian GW scenarios[3]								
	Rustad	1.9	Silty clay loam	13	60	27	6.6	1.3
	Heia	2.2	Sandy loam	5	30	65	6.4	1.4

In 2011 the first initiative was taken to harmonize groundwater requirement for pesticide leaching scenarios in the Northern Zone (Kruskops, 2015). The main goal of this project was to find a worst case ranking of the scenarios. The result showed that the ranking was highly dependent on the pesticide properties like DT50, Koc and 1/n and could not be generalized, and a data gap was identified: What is the actual extent of geographical representativeness of the existing modelling scenarios in the Northern Zone. A new project was initiated in 2014 to follow up to assess the representativeness and protectiveness for the existing modelling scenarios. Representativeness was assessed by analysing and comparing the soil and climate characteristics of the relevant data (Burns et al., 2015).

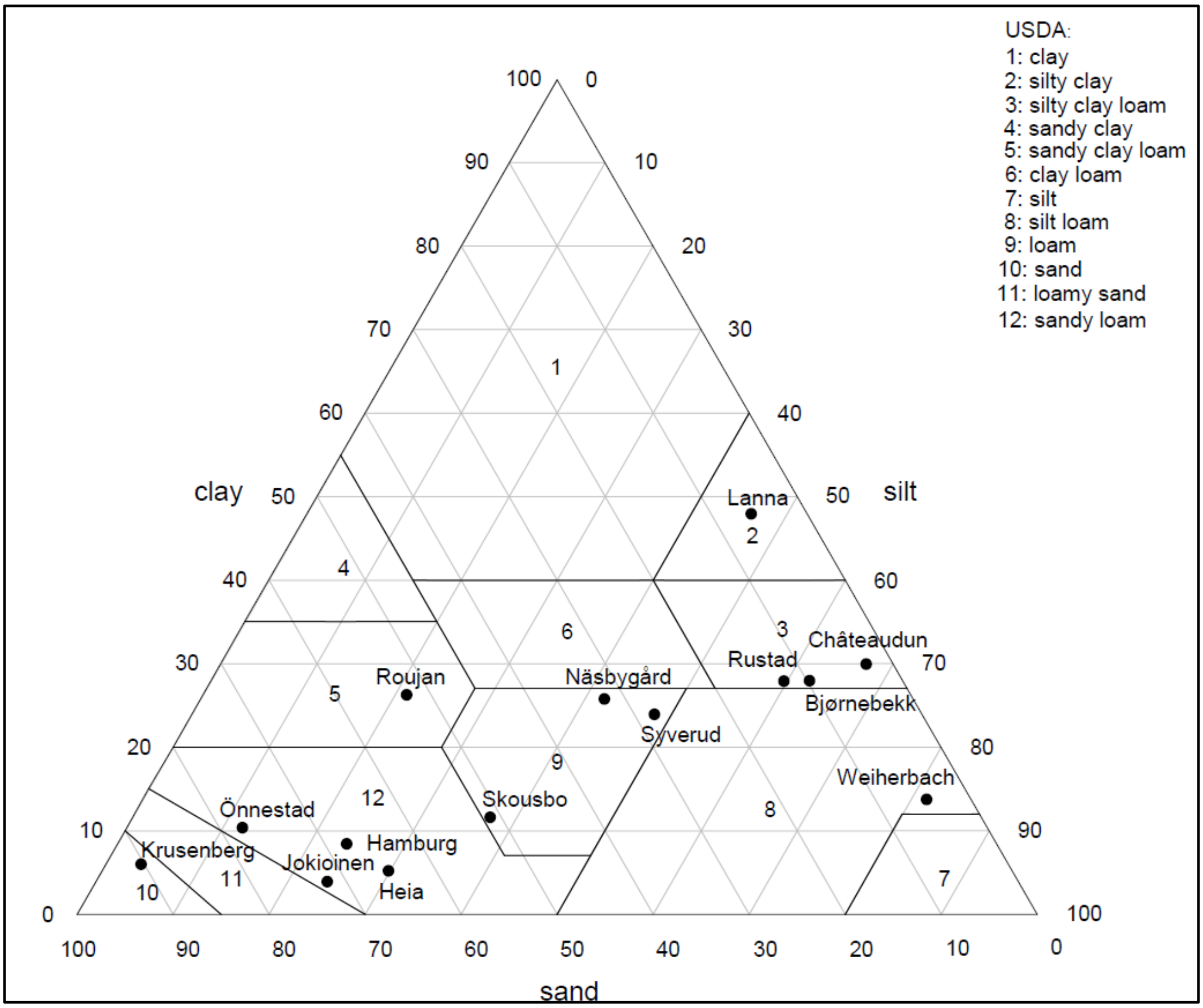


Figure 5.3-2 Soil textural classes in the top layer (0-20 cm) for different sites, according to USDA (USDA, 1996).

5.3.1 Soil Scenarios

The first part of the project was to analyse the representativity of the soil (Table 5.3.1-1). To do that the Focus scenarios, Hamburg and Jokioinen and the national scenarios for Sweden (Krusenberg, Önnestad and Näsbygard), Denmark (Karup, Langvad) and Norway (Rustad and Heia) had to be translated to FOOTPRINT soil types (FST). This allowed for a quantification of the representativity of the FOCUS/national soil scenarios for the agricultural areas of each country and the whole northern zone calculated as a percentage of the agricultural area. Unfortunately, the soil map of Europe did not include Norway and calculation of percentage of the agricultural area in Norway was not possible. But the soils from Norway was translated and classified according to the Footprint database of EU. The soil from Heia was translated to the hydrological group Y, which is slowly permeable substrate and shallow perched water table belonging to the same group as soil type as Langevad from Denmark, and Näsbygard from Sweden (Table 5.3.1-1).

Table 5.3.1-1 Results of the translation of the soils that constitute the national scenarios to FSTs (Burns et al., 2015).

Country	Scenario name	FSTmap	FSTmodelled
Norway	Rustad	Y34ih	Y34ih
	Heia	Y22n	Y22n
Denmark	Karup	L11n	L11n
	Langvad	Y22n	Y22n
Sweden	Krusenberg	Y14i	Y14i
	Önnestad	L11n	L11n
	Näsbygard	Y22n	Y22n
FOCUS GW	Jokioinen	O11p	O11p
	Hamburg	O11n	O11n

The analysis of the representativity of the soil types from the different location showed that the hydrological group of the soil at Heia represented totally 27% of the Northern zone and especially in Latvia more than 75% of the agricultural area was included. Analysing the representativity of soil according to the texture of the top and sub soil, the Heia soil represented 34 % of the agricultural land within the Northern zone. Especially Sweden was well represented with 59 % of the area.

The soil from Rustad was also classified in the same hydrological group, but the texture of the top and sub soil had almost no representation within the other Nordic and Baltic countries, only in the FOCUS Chateaudun belong to the same part of the soil triangle (Figure 5.3-2).

5.3.2 Climate

Analysis of the representativity of the climate scenarios was also performed by Burns et al. (Burns et al., 2015), concluding that within the Northern zone the different climate scenarios were less representative than the soil used in the FOCUS and national scenarios. The climate scenario used for this exercise was climate for with average annual temperature 5.7 ºC and 851mm precipitation. Climatically Norway falls into 4 of the FOCUS regions (Figure 1-2).

The eastern part of Norway corresponds to the mid-part of Sweden and parts of Finland. Sothern part of Norway correspond more to Denmark and south of Sweden. South western part of Norway corresponds more to the English weather.

The Footways report generally conclude that the national and the FOCUS soil and climate scenarios were found to be variably, but overall poorly representative of the true soil and climate conditions of the northern zone (Burns et al., 2015). Representativeness was worse for the climate scenario than for the soil scenarios, but the combination of both was even poorer. Representativity in the Footways report is defined as percentage of the area in the region with same soil type and climate. This is truly different from the scenario modelling representativity defined in the FOCUS groundwater scenario which is more describing the protectiveness.

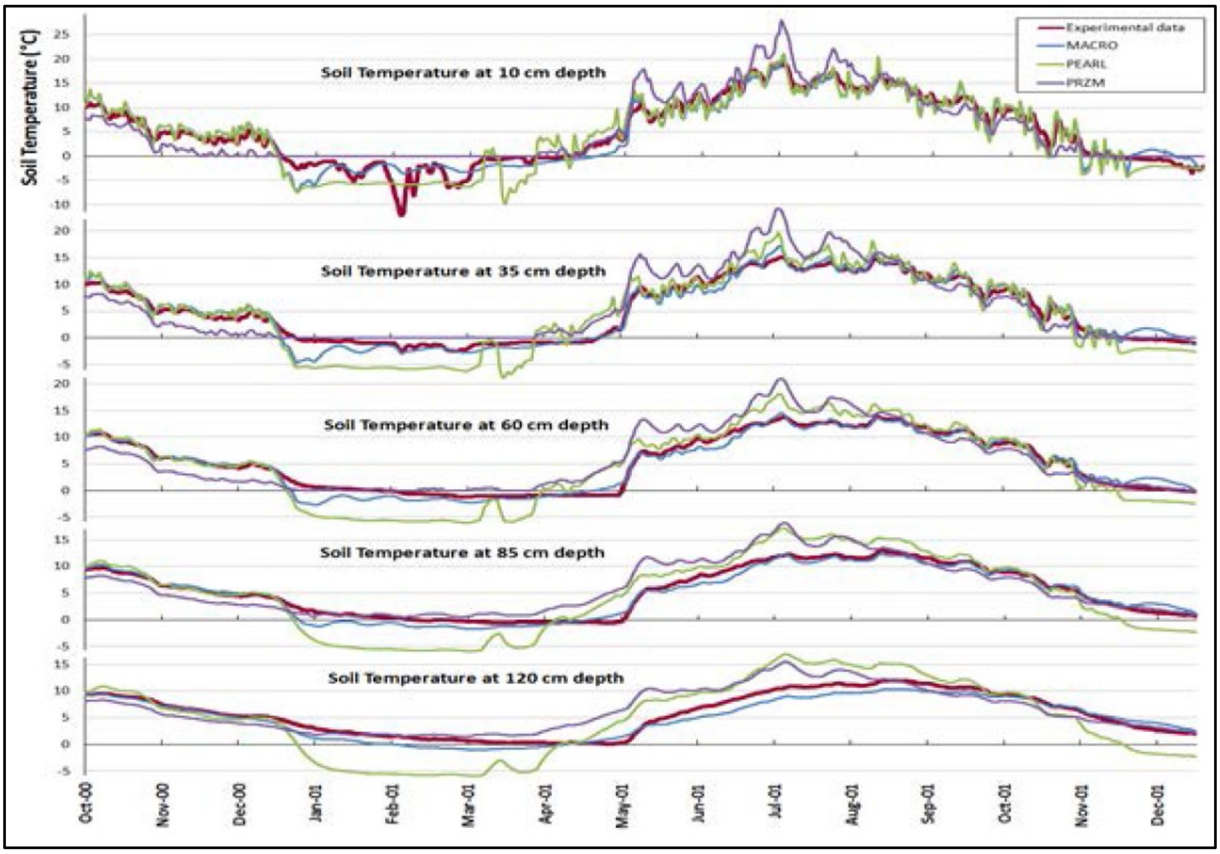


Figure 5.3.2-1 Temperature simulation profiles at Grue (Gomez, 2012)

From a field study at Grue (Gomez, 2012), most of the models were not able to simulate the winter condition and especially temperature and soil moisture condition in winter and spring was difficult (Figure 5.3.2-1). Especially the snowmelt situation in the end of April is difficult to simulate, which represent a period of increased transport of water downward to the groundwater.

Figure 5.3.2-2 Soil water content simulation profiles at Grue (Gomez, 2012)

To compare different scenarios, simulations with FOCUS-MACRO were performed for the FOCUS scenario Châteaudun and the Swedish, Danish and Norwegian scenarios (Bolli et al., 2011). Norwegian endpoints from field studies and EU endpoints (Table 5.3.2-1) have been used, either combined with a Norwegian climate file from southeast of Norway or climate files belonging to each scenario. These were the only input parameters which were changed in the simulations. For other parameters in the model, FOCUS default values have been used. The scenario from Châteaudun was only simulated with the scenario specific climate file, because of problems when using another climate file. The general parameters in the simulations like application rate, application date, interception and crop were the same for all scenarios. The chemical properties like molecular mass, vapour pressure and water solubility were also the same for all scenarios.

Table 5.3.2-1 Input pesticide parameters for groundwater modelling for the estimation of Predicted Environmental Concentrations (PEC) for propiconazole, metalaxyl and isoproturon.

	Norwegian endpoints	EU endpoints
Propiconazole		
DT50$_{soil}$ (d):	201 (geomean, n=2)	72
K$_{foc}$:	984	382
1/n:	1.13	0.90
Metalaxyl		
DT50$_{soil}$ (d):	64 (geomean, n=2)	36 (median, n=7)
K$_{foc}$:	20	162
1/n:	0.90	0.90
Isoproturon		
DT50$_{soil}$ (d):	13 (geomean, n=2)	11.9 (geomean, n=2)
K$_{foc}$:	95	104
1/n:	0.96	0.90

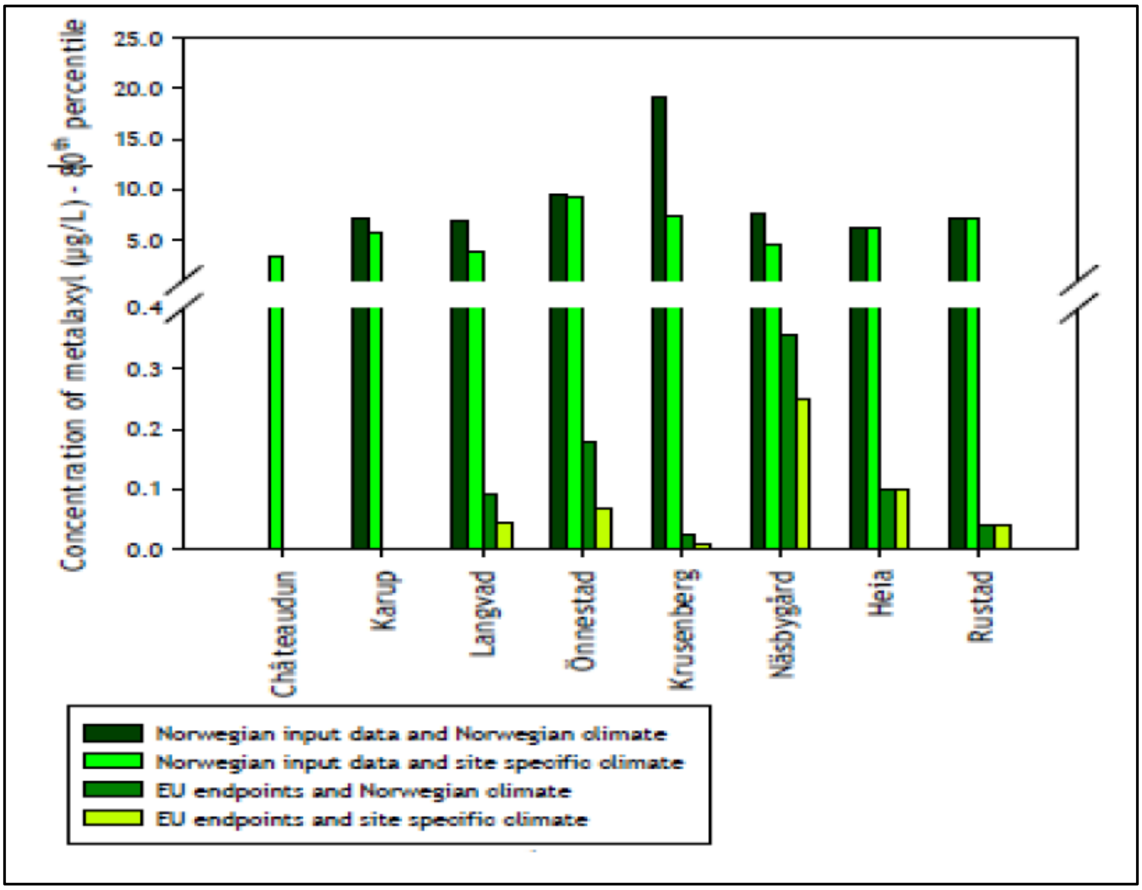


Figure 5.3.2-3 Leaching of metalaxyl (80th percentile) from different scenarios, using Norwegian endpoints and EU endpoints together with a Norwegian climate file and scenario specific climate files simulated with FOCUS-MACRO (Bolli et al., 2011).

Running the different scenarios with EU-endpoints with climate files from Norway gave possibilities to look at the direct effect of precipitation on transport and temperature on leaching. The 80th percentile for almost all of the sites (locations) showed increased leaching for all pesticides tested (propiconazole, isoproturon and metalaxyl) when using Norwegian climate compared to simulations using site specific climate.

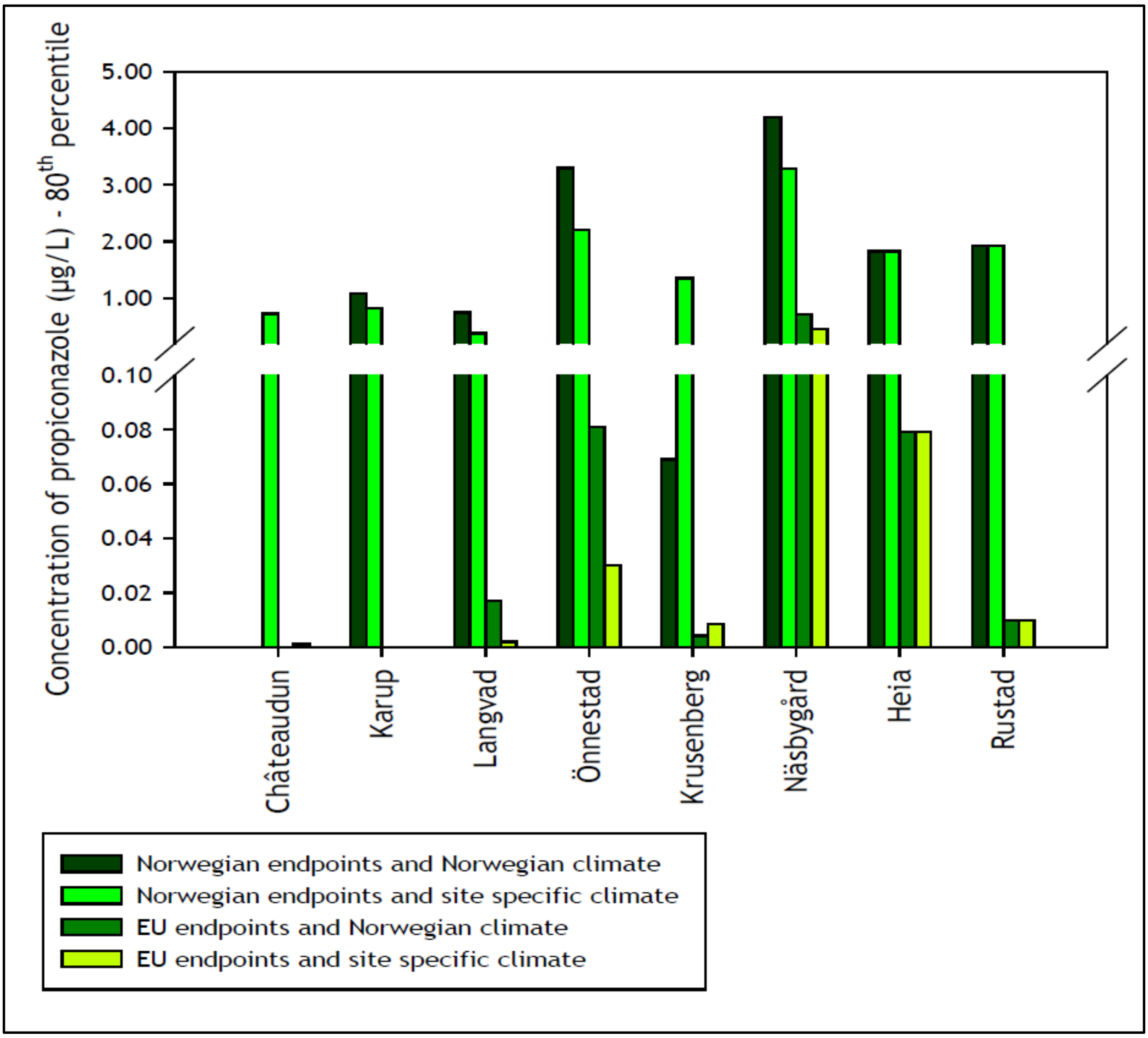


Figure 5.3.2-4 Leaching of propiconazole (80th percentile) from different scenarios, using Norwegian endpoints and EU endpoints together with Norwegian climate file and scenario specific climate files simulated with FOCUS-MACRO (Bolli et al., 2011).

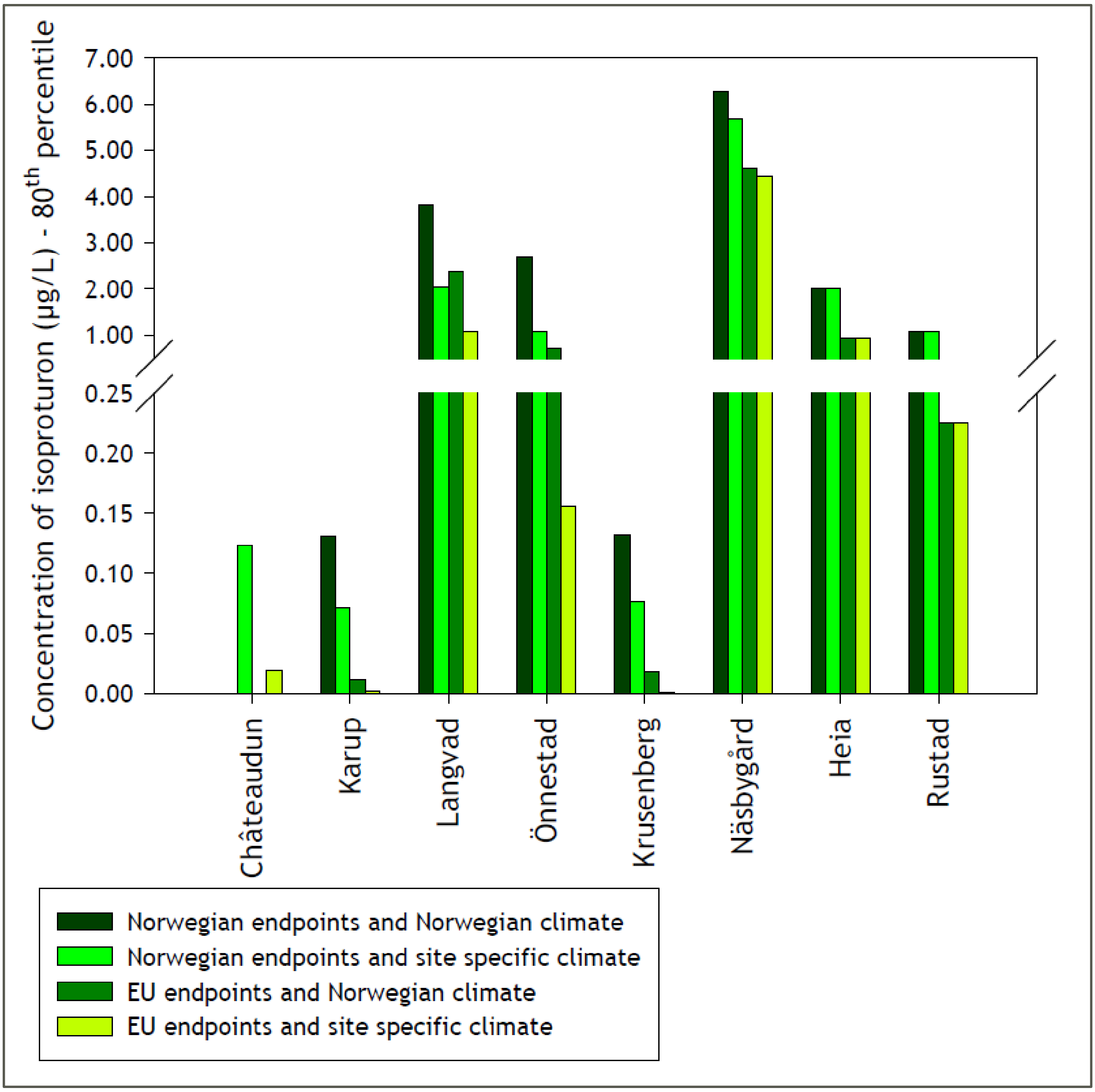


Figure 5.3.2-5 Leaching of isoproturon (80th percentile) from different scenarios, using Norwegian endpoints and EU endpoints together with a Norwegian climate file and scenario specific climate files simulated with FOCUS-MACRO (Bolli et al., 2011).

However, the direct effect of the Norwegian climate was relatively low especially for the mobile fungicide metalaxyl. Climate will indirectly affect the exposure of pesticide by its effect on soil quality (properties) and moisture which influence degradation and especially microbial activity but also sorption and transport. The indirect effects of climate were expressed by using the specific Norwegian endpoints for sorption and degradation in the other scenarios. This increases the leaching for all sites. The leaching of metalaxyl was most affected by the Norwegian endpoints, while the leaching of isoproturon was less affected. When the Norwegian climate file and Norwegian endpoints were added, the leaching of all pesticides increased even more (Bolli et al., 2011).

5.3.3 Protectiveness

Burns et al. (Burns et al., 2015) underline the definition of representativeness of a scenario, refers to agricultural areas with similar soil and climate conditions. Protectiveness of a scenario refers to the agricultural area which has similar or better-case soil/climate conditions. Protectiveness of a scenario can be viewed as a specific point on a spatial cumulative distribution function (CDF) of worst-case-ness of a country or zone. What matters for risk assessment purposes is not so much representativeness, but protectiveness, i.e. where the scenario fits in relation to the whole spectrum of risks.

In the following exercise, Burns et al. (Burns et al., 2015) were modelling leaching of pesticides to groundwater, using the Proziris platform with spatial dataset on soil, climate and land-use for the Nordic Zone. The spatial resolution of land-use database was $1km^2x1km^2$. Eighteen substances were used in in the modelling with EU standard parametrisation and Danish substance parametrisation with lower Kd and slower degradation. Eight indicators (ie. 80th percentile of annual PECgw etc.) were calculated from the model scenarios. The assessment of the protectiveness was achieved by comparing the results obtained from the individual national modelling approaches to the CDFs generated with the Proziris modelling. Indicator results obtained for the FOCUS/national scenario modelling were plotted over the Prozitris spatial cumulative distribution functions (CDFs). This was done for the various leaching endpoints.

The protectiveness values for the two national scenarios from Heia (HE-220-MACRO) and Rustad (RU-220-MACRO) were quantified using CDF of the risk indicator "second highest annual predicted groundwater concentration". The percentile read from the graph (Figure 5.3.3-1) is close to 90 percentile for the two scenarios. Compared to the spatial modelling by Proziris within the area the mean protectiveness of the national scenarios is sufficiently high for lower tier regulatory leaching assessments for the Northern zone. Because of the uncertainty of the protectiveness of the scenario-based approach, according to Burn et al (2015), single scenario in regulatory modelling is not sufficient. At the zonal level all seven national (Danish, Swedish and Norwegian) scenarios should be simulated. In contrast to lower-tier assessments, in higher-tier assessments the exceedance area percentage of the 0.1 µg/L limit has to be exactly known. Because their protectiveness varies, for higher tier-simulations, a GIS-based, fully spatially probabilistic approach such as Proziris should be used (Burns et al., 2015). Such tools are also requested from the regulatory bodies. This is a way to include variability of soil and climate, but endpoints for sorption and degradation still have to be produced for representing the zone.

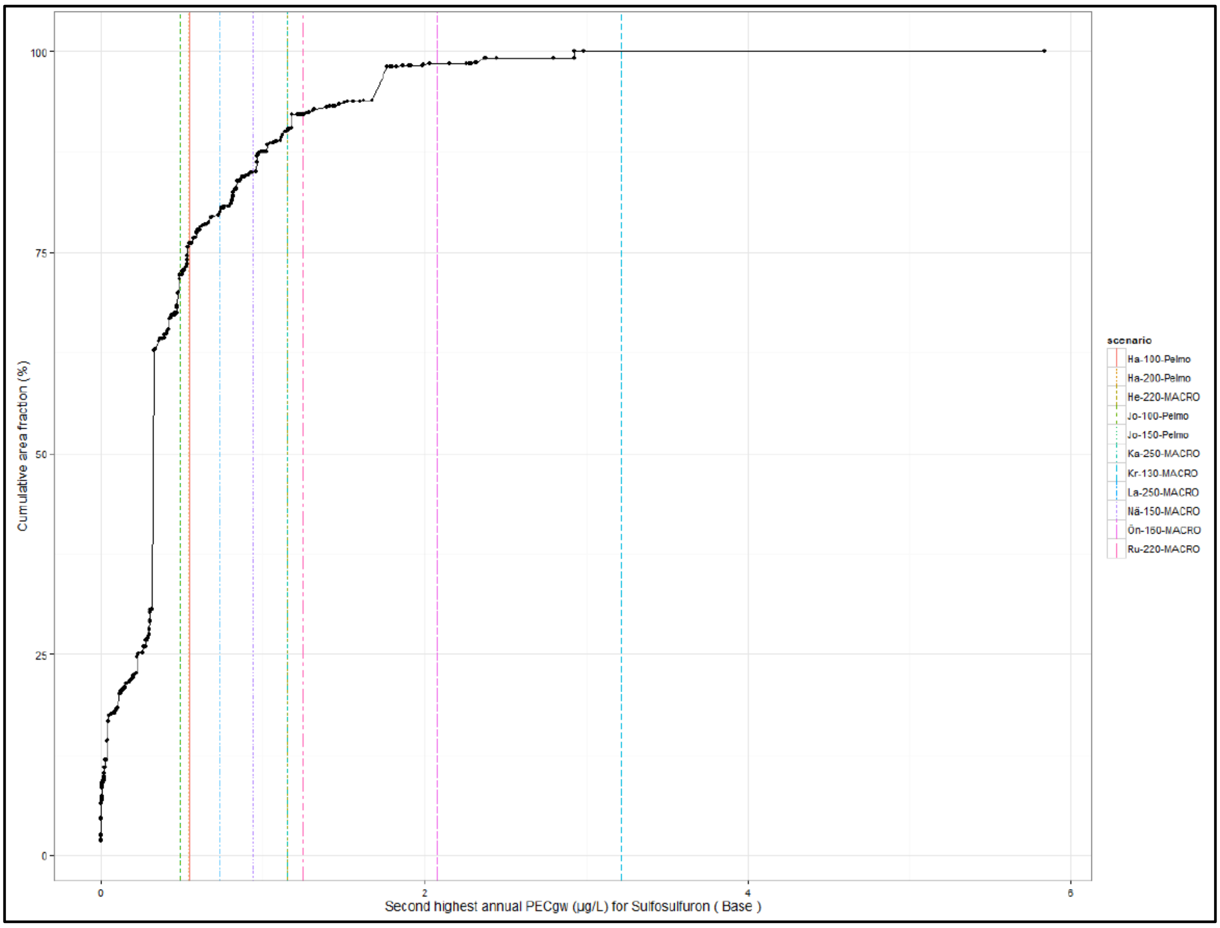


Figure 5.3.3-1 Example of how the protectiveness of the FOCUS / national scenarios were quantified using a CDF of the risk indicator second highest (80th percentile) annual PECgw for Sulfosulfuron generated by the Proziris platform. The legend presents a code that describes three different types of information: scenario name, leaching evaluation depth, and model used. Combinations of scenario, output depth and model with values outside the range of the CDF are not shown in the graph (Burns et al., 2015).

5.3.4 Variability of sorption and degradation as input for modelling

Pesticide fate and leaching can vary a lot in the different soil types. In the Proziris platform using spatial simulation with different soil, climate and land cover/land use, the variability of pesticide behaviour according to sorption and degradation is not taken into account. Here the EU endpoints or fixed endpoints are used for the spatial simulations scenarios avoiding effects of these parameters. For the FOCUS dummy substances the Danish regulatory risk assessment are using a "safety" factor by multiplying the Koc values with 0.65, Freundlich 1/n by 1.039 and DT50 in soil by 1.54 is used.

Different sour of uncertainties in pesticide fate modelling has been investigated by several authors and reviewed by Dubus et al. (Dubus et al., 2003). This article reviews different sources of uncertainty associated with pesticide fate modelling in general, but put emphasis on simulation of pesticide leaching through soil. In this review the uncertainty terminology included variation, variability, heterogeneity, approximation etc. and used uncertainty in the widest sense of the word. Relevance for this evaluation of field data in modelling is uncertainty in the primary data including spatial and temporal variability of environmental variables as the capacity of soils for sorption and degradation play an important role within the context of pesticide fate modelling (Boesten, 1991; Dubus et al., 2003).

Variability in the sorption distribution coefficient Kd can generally be reduced by normalizing it to the organic carbon content (Hamaker and Thompson, 1972), but the variability of the resulting Koc often remains considerable. Some authors have observed that normalization of Kd to organic carbon fails to reduce variability (Beck et al., 1996; Elabd et al., 1986), and especially emphasize the Koc concept is developed and established from non-ionic compounds (Hamaker and Thompson, 1972). According to Dubus and coworkers (Dubus et al., 2003) one of the largest sources of error in modelling is inadequate use of Koc and the parametrization regardless of ionization status for the pesticide. The Koc approach is invalid for ionisable compounds (Wauchope et al., 2002).

In the sorption data from the Nordic reference soils (Greve et al., 1998) the Kd varies from 1.5 to 166 for atrazine measured in 13 soil types from the Nordic countries. Normalizing the Kd to organic carbon content the Koc varies from 94 to 472. For atrazine the strong sorption seems here to be influenced by low pH and acidic soil with high content of organic matter. The correlation (R^2) between pH and Kd for atrazine was 0.69, as the correlation between total carbon and Kd was 0.78. The ionic pesticide 2,4-D in general has low sorption in most agricultural soils. Sorption to soil with low content of total organic C (1.1-2.4) and pH between 5.4 to 6.3 the Kd is 0.4 - 1.4. Including the soil with high content of organic carbon the range increases to 0.4 to 265. Normalizing Kd for the pesticide 2,4-D the Koc range increases even more as 2,4-D is an an-ionic substance.

Spatial variability of degradation has received less attention, but soil types with the same texture might have different risk of leaching of pesticide to groundwater. Because of slowly permeable soil deeper in the soil profile. Because of the heterogeneity of soil, risk of leaching can vary within short distances within the same field. This has been illustrated by combining GIS with model simulation with MACRO (Figure 5.3.4-1 and Figure 5.3.4-2). From a model study in Sweden a small part (1%) of the catchment contributed to the main part (70 %) of the diffuse pollution at the farm. From other areas, it has been reported that from 1 to 17 % of the area has contributed to 90 % of the pollution (Lindahl et al., 2005).

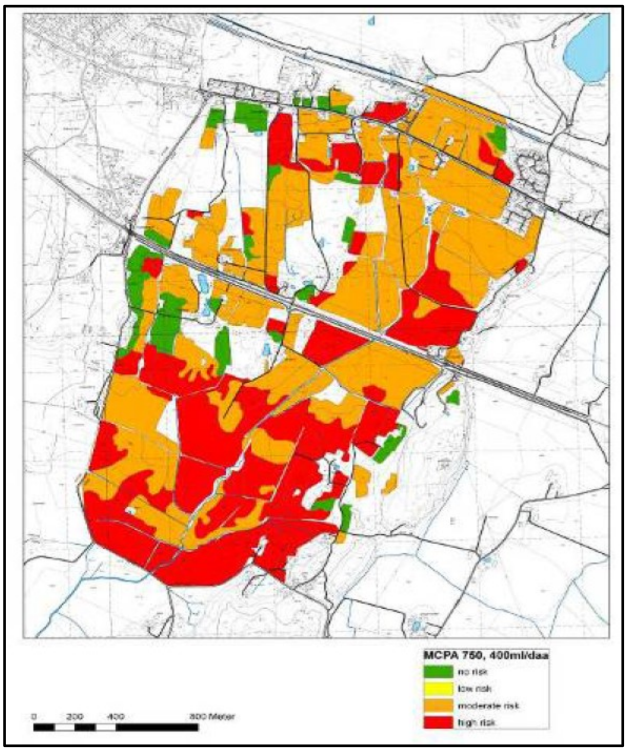


Figure 5.3.4-1 Risk of leaching to groundwater of MCPA (Eklo et al., 2009).

Figure 5.3.4-2 Risk of leaching of Primus to groundwater (Eklo et al., 2009).

5.3.5 Conclusion groundwater

The soil and climate conditions in the national and FOCUS scenarios were found to be poorly representative for the Northern zone (Kruskops, 2015). Spatial soil structure was found to be variable, but representativeness of the climate scenarios was worse than for the soil scenarios. The combination of soil and climate was even poorer. When combining soil and climate characteristic, only about 1.1 % of the agricultural area in the Northern zone was represented by all the group of groundwater scenarios used in the Northern zone. Some of the climate scenarios are old and do not reflect the climate change (Burns et al., 2015).

Generally a relative high protectiveness for the national scenarios of MACRO was demonstrated, but varied between substances and implies that the uncertainty will decrease with increasing numbers of scenarios. To rely upon one single scenario in regulatory modelling cannot be recommended. Combinations of national scenarios from Önnestad, Langvad and Rustad are suggested as a possible option, which ensures variable soil conditions and climate representativity. Burns et al. (Burns et al., 2015) conclude that for higher tier-simulations, a GIS-based, fully spatially probabilistic approach such as Proziris should be used. The approach to multiply the three main factors for leaching; KOC, 1/n and T1/2 by safety factors, as used by Denmark for the FOCUS dummy substances, is a pragmatic solution to achieve a protection that is adequate for the Norwegian conditions.

5.4 Leaching and runoff to surface water (PECsw)

The surface water exposure estimation of the risk assessment process is performed according to a stepwise tiered approach within EU. The first step is an extreme worst case loading. If this step is considered not safe, the step 2 is necessary. This step assumes sequential application pattern, taking into account degradation of the substance between the applications. If this step still is considered as unsafe use, further work with step 3 is necessary. In step 3 exposure simulations using a set of 10 scenarios are used, representing realistic worst case with four runoff scenarios and six drainage scenarios (Figure 5.4-1). These predicted concentrations are compared with toxicity parameters for water living organisms and if a risk is still identified, the procedure proceeds to step 4, exposure estimation. This step includes a variety of refinements and mitigation measures.

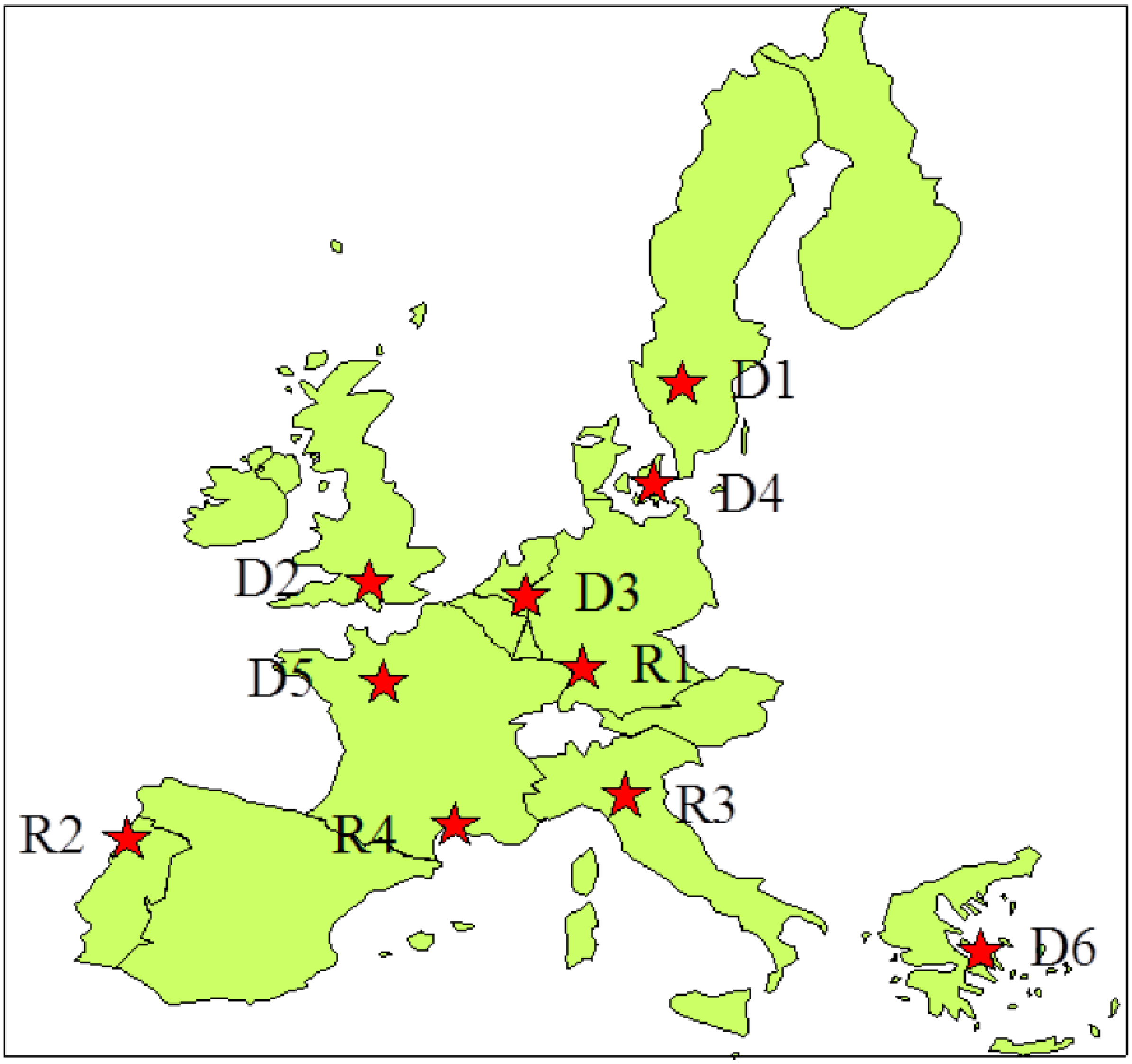


Figure 5.4-1 For the ten surface water scenarios realistic worst case according to climate, slope and soil are identified (FOCUS, 2001).

5.4.1 Soil scenarios

The soil texture scenarios at the FOCUS and Norwegian surface water scenarios are shown in table 5.4.1-1. The surface runoff scenario Roujan has approximately the same clay content as Bjørnebekk, but has a lower content of silt and a higher content of sand. Weiherbach has almost the same sand content as Bjørnebekk, but with a lower content of clay and a higher content of silt.

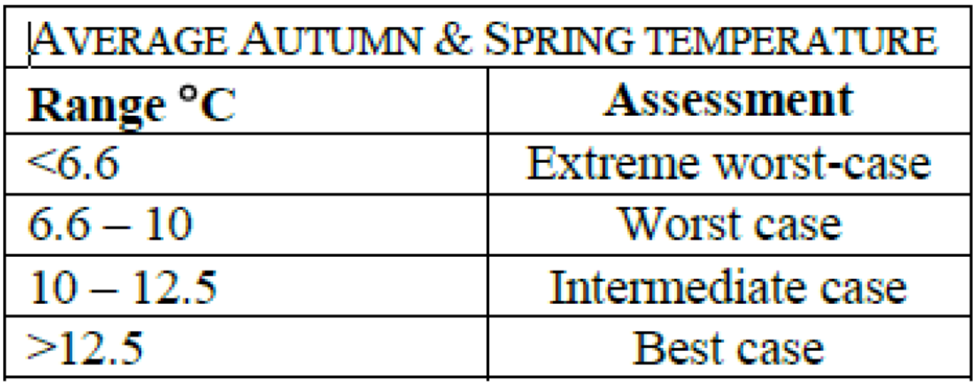
Table 5.4.1-1 Soil texture of the soil from surface water scenarios in Europe and Norway.

Source	Representative field site	Organic carbon %	Texture class	Clay [1] %	Silt [1] %	Sand [1] %	pH	Bulk density g cm^{-3}
Focus SW Scenario[2]								
D1	Lanna	2.0	Silty clay	47	46	7	7.2	1.35
D2	Brimstone	3.3	Clay	54	39	7	7.0	1.20
D3	Vredepeel	2.3	Sand	3	6	91	5.3	1.35
D4	Skousbo	1.4	Loam	12	37	51	6.9	1.48
D5	La Jailliere	2.1	Loam	19	39	42	6.5	1.55
D6	Váyia, Thiva	1.2	Clay loam	30	34	36	7.5	1.43
R1	Weiherbach	1.2	Silt loam	13	82	5	7.3	1.35
R2	Valadares, Porto	4.0	Sandy loam	14	19	67	4.5	1.15
R3	Ozzano, Bologna	1.0	Clay loam	34	43	23	7.9	1.46
R4	Roujan	0.6	Sandy clay loam	25	22	53	8.4	1.52
Norwegian SW Scenarios[3]								
	Bjørnebekk	1.5	Silty clay loam	26	64	9	6.0	
	Syverud	3.1	Loam/silt loam	27	47	26	5.5	

5.4.2 Climate

In terms of average temperature in most of Norway falls into the "worst-case" or "extreme worst case" categories as defined by (FOCUS, 2001). (See table 5.4.2-1). Only along the southwest coast the climate is warmer than "worst case". The Norwegian scenario from Syverud still belongs to the extreme worst case because of cold winter and spring conditions.

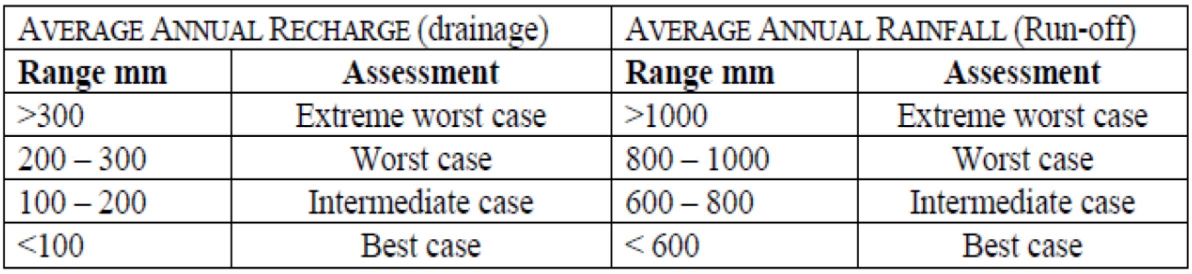
Table 5.4.2-1 Climatic temperature classes in the agricultural scenarios.



AVERAGE AUTUMN & SPRING TEMPERATURE	
Range °C	**Assessment**
<6.6	Extreme worst-case
6.6 – 10	Worst case
10 – 12.5	Intermediate case
>12.5	Best case

In the FOCUS document (FOCUS, 2001), the climatic differentiation for agricultural drainage and runoff scenarios have been classified according to recharge and average annual rainfall (Table 5.4.2-2). Most of the agricultural areas in Norway except for the south eastern part with less precipitation falls into the extreme worst case category (Table 5.4.2-5).

Table 5.4.2-2 Climatic classes for differentiating agricultural drainage and runoff scenarios (FOCUS, 2001).



AVERAGE ANNUAL RECHARGE (drainage)		AVERAGE ANNUAL RAINFALL (Run-off)	
Range mm	**Assessment**	**Range mm**	**Assessment**
>300	Extreme worst case	>1000	Extreme worst case
200 – 300	Worst case	800 – 1000	Worst case
100 – 200	Intermediate case	600 – 800	Intermediate case
<100	Best case	< 600	Best case

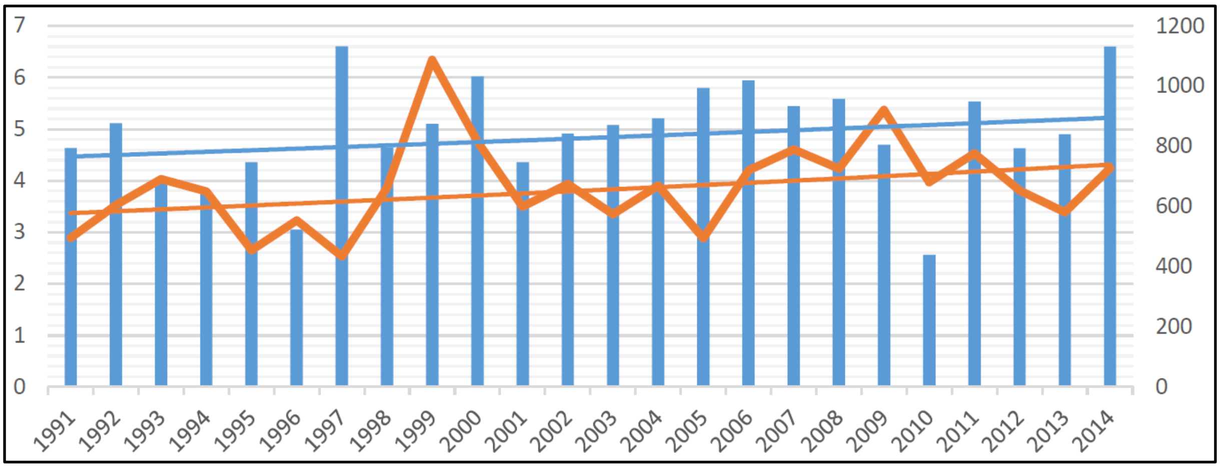
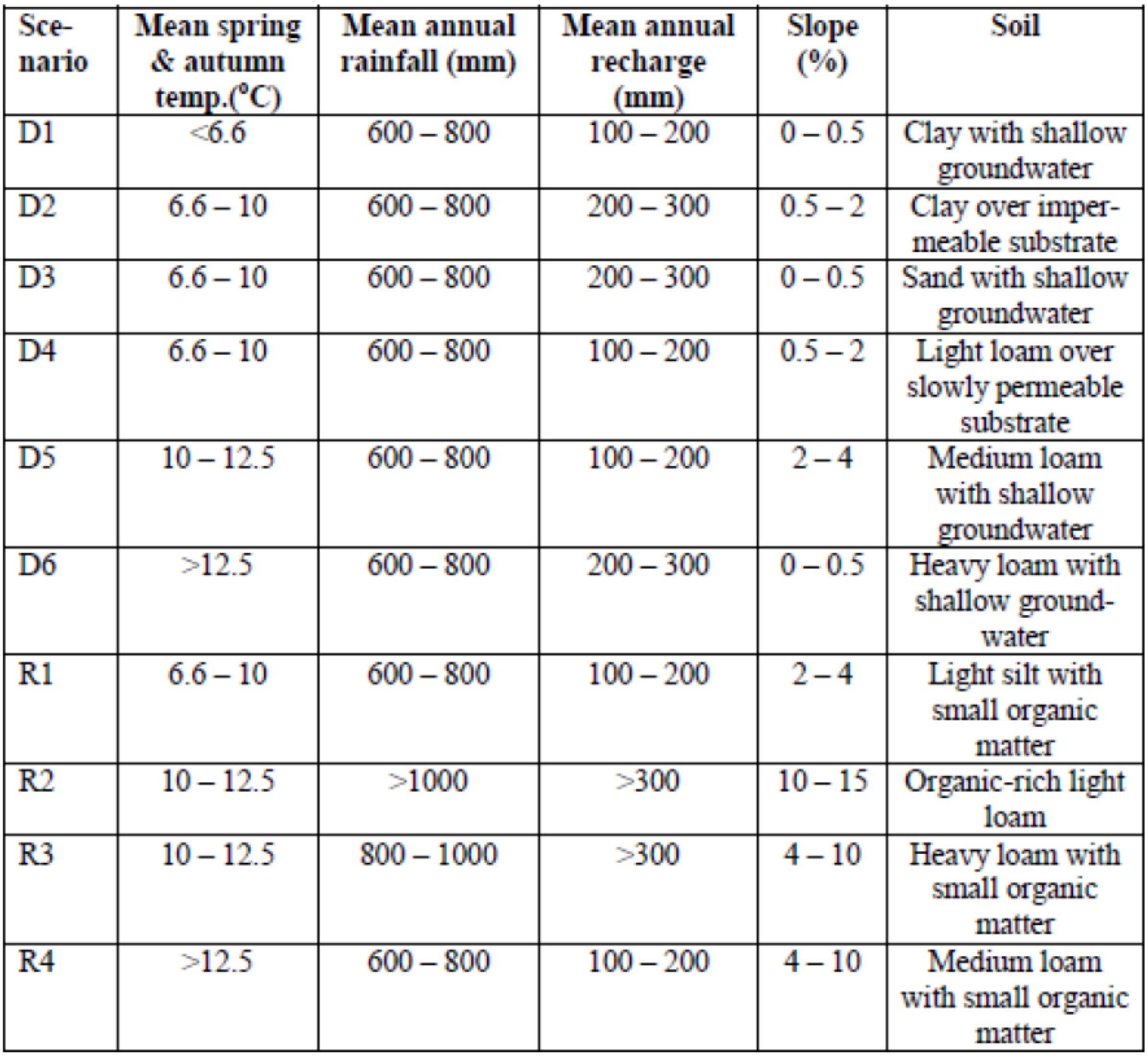


Figure 5.4.2-1 Average annual temperature and precipitation at Apelsvoll (North Eastern Norway) from 1991-2013.

In most of agricultural regions in Norway, both temperature and precipitation has changed and mainly increased like the situation at Apelsvoll (Figure 5.4.2-1), which can represent the climate conditions of the drainage scenarios (Syverud and Bjørnebekk). Compared to the climatic classes in the FOCUS scenarios (Table 5.4.2-2), the drainage scenario is moving from intermediate to worst case of annual rainfall.

The Agro-environmental characteristics of the FOCUS surface water scenarios are listed in table 5.4.2-3

Table 5.4.2-3 Agro-environmental characteristics of the surface water scenarios.



Scenario	Mean spring & autumn temp.(°C)	Mean annual rainfall (mm)	Mean annual recharge (mm)	Slope (%)	Soil
D1	<6.6	600 – 800	100 – 200	0 – 0.5	Clay with shallow groundwater
D2	6.6 – 10	600 – 800	200 – 300	0.5 – 2	Clay over impermeable substrate
D3	6.6 – 10	600 – 800	200 – 300	0 – 0.5	Sand with shallow groundwater
D4	6.6 – 10	600 – 800	100 – 200	0.5 – 2	Light loam over slowly permeable substrate
D5	10 – 12.5	600 – 800	100 – 200	2 – 4	Medium loam with shallow groundwater
D6	>12.5	600 – 800	200 – 300	0 – 0.5	Heavy loam with shallow groundwater
R1	6.6 – 10	600 – 800	100 – 200	2 – 4	Light silt with small organic matter
R2	10 – 12.5	>1000	>300	10 – 15	Organic-rich light loam
R3	10 – 12.5	800 – 1000	>300	4 – 10	Heavy loam with small organic matter
R4	>12.5	600 – 800	100 – 200	4 – 10	Medium loam with small organic matter

In most of the European scenarios, annual rainfall is below 800 mm. Compared to the agro-environmental conditions in south-eastern Norway this is within the range of normal precipitation.

Normally more hilly areas are used for agricultural purpose in Norway, only R2 is comparable. The soil and climatic conditions and representability is discussed in a previous chapter. In the FOCUS document (FOCUS, 2001) the distribution of the different scenarios are discussed, but Norway is not included in the maps and information of the Northern zone is often lacking.

For the Northern zone Predicted concentration in surface water (PEC_{sw}), is to be calculated with the FOCUS STEP3 scenarios in accordance to the country specific requirements. For

Norway all ten scenarios are required and with relevant mitigation measures (step4) (See Table 5.4.2-4).

Table 5.4.2-4 Country specific requirements for FOCUS scenarios considered in the assessment of surface water and sediment exposure (GUIDANCE DOCUMENT, 2014).

Country	**Scenarios**									
	D1	**D2**	**D3**	**D4**	**D5**	**D6**	**R1**	**R2**	**R3**	**R4**
Denmark			X	X						
Estonia	X		X	X			X			
Sweden	X			X			X			
Norway	X	X	X	X	X	X	X	X	X	X
Lithuania	X		X	X			X			
Latvia	X		X	X			X			
Finland	X			X			X			

Step 3 requires the use of deterministic models such as PRZM, MACRO and TOXSWA, and because of lacking relevant information for Norway a new tool WISPE was developed for surface runoff calculations. PRZM was implemented in a new software WISPE (World Integrated System for Pesticide Exposure) (Cheplick et al., 2012), which was a collaboration between Waterborne Environmental Inc., Norwegian Food Safety Authority and Bioforsk. The WISPE model is not in use for regulatory purpose for the moment.

Since 2001, four different sites Heia, Rustad , Syverud and Bjørnebekk have been involved in the study with the pesticides; isoproturon, metalaxyl and propiconazol (Bolli et al., 2013; Bolli et al., 2011; Eklo et al., 2009; Eklo et al., 2008; Haugen et al., 2002). Data from these studies has been used for calibration and validation of the two models MACRO (Jarvis, 1994) and PRZM (Pesticide Root Zone Model) (Suárez, 2005). PRZM simulates time-varying hydrologic behaviour on a daily time step, including physical processes of runoff, infiltration, erosion, and evapotranspiration. The chemical transport component of PRZM calculates pesticide uptake by plants, surface runoff, erosion, decay, vertical movement, foliar loss, dispersion, and retardation. The architecture of WISPE allows seamless executions of several environmental fate and transport models including PRZM and EXAMS. EXAMX calculates the concentrations in the water body following the TOXWA description (FOCUS, 2001) calculating exposure of pesticides in ditch, pond and streams.

In a modelling exercise with WISPE scenarios from Syverud and Bjørnebekk in Norway was compared with the FOCUS runoff scenarios: R1-R4 from Weiherbach, Porto, Bologna and Roujan, and the drainage scenarios: D1, D3, D4, D5 and D6 at respectively Lanna, Vredepeel, Skousbo, La Jailliere and Thiva. All scenarios were parameterized according to the Generic guidance for FOCUS Surface Water Scenarios (FOCUS, 2001). All scenario simulations were run with data on soil, topography and climate specific for each of the sites. For the pesticide properties, EU-end points and application rates were used for all simulations (Table 5.4.2-5). The pesticides used in the modelling were metalaxyl-m, propiconazole, diflufenican, fluropyran and seven artificial test compounds. The global max concentrations of the pesticides were calculated in ditch, pond and stream giving the overall highest concentration. Comparing FOCUS and WISPE there was no big differences between mobile pesticides, as demonstrated for metalaxyl in Figure 5.4.2-2.

Table 5.4.2-5 Parameters and application rates used for the simulation with FOCUS and WISPE

Substance	**MW**	**Vapour pressure (Pa)**	**Water solubility (mg/L) 20 °C**	**DT50 soil, days lab 20 °C**	**DT50 water days**	**DT50sed days**	**Koc**	**1/n**	**Plant Uptake Factor**	**Appl. rate, g a.s./ha**
Metalaxyl-M	279,3	0,0033 (25°)	26000 (25 °C)	6,2	47,5	1000	40,0	0,93	0,5	1x363
Propiconazole	342,2	5,6x10^{-5} (25 °C)	150	48,1	6	1000	1185	0,9	0,5	2x125
Diflufenican	394	4,25x10^{-6} (25 °C)	0,05	143,2	175	1000	3417	0,917	0	1x120
Fluopyram	396,7	1,2x10^{-6} (20 °C)	16	118,8	1000	1000	278,9	0,827	0	2x250

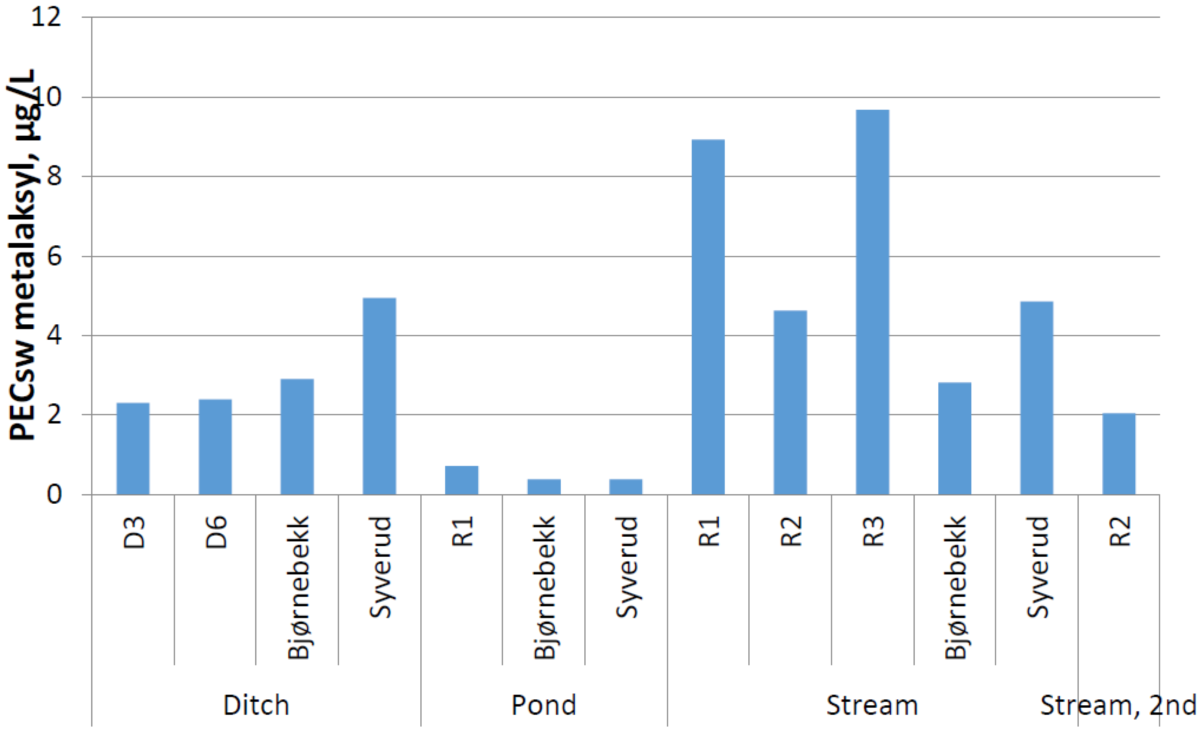


Figure 5.4.2-2 Predicted concentration of metalaxyl in ditch, pond and stream using the FOCUS SWASH scenarios and the Norwegian scenarios using WISPE.

For two of the more strongly sorbed pesticides, propiconazole and diflufenican, the concentration in the water phase in pond simulations is higher for the Norwegian sites than in the FOCUS scenarios (Figure 5.4.2-3 and 5.4.2-4). However, for all scenarios the concentrations in ponds are much lower than in ditch and stream simulations. The ditch and stream simulations do not differ much between the Norwegian and FOCUS scenarios. For fluopyran, this applies also to the pond simulations.

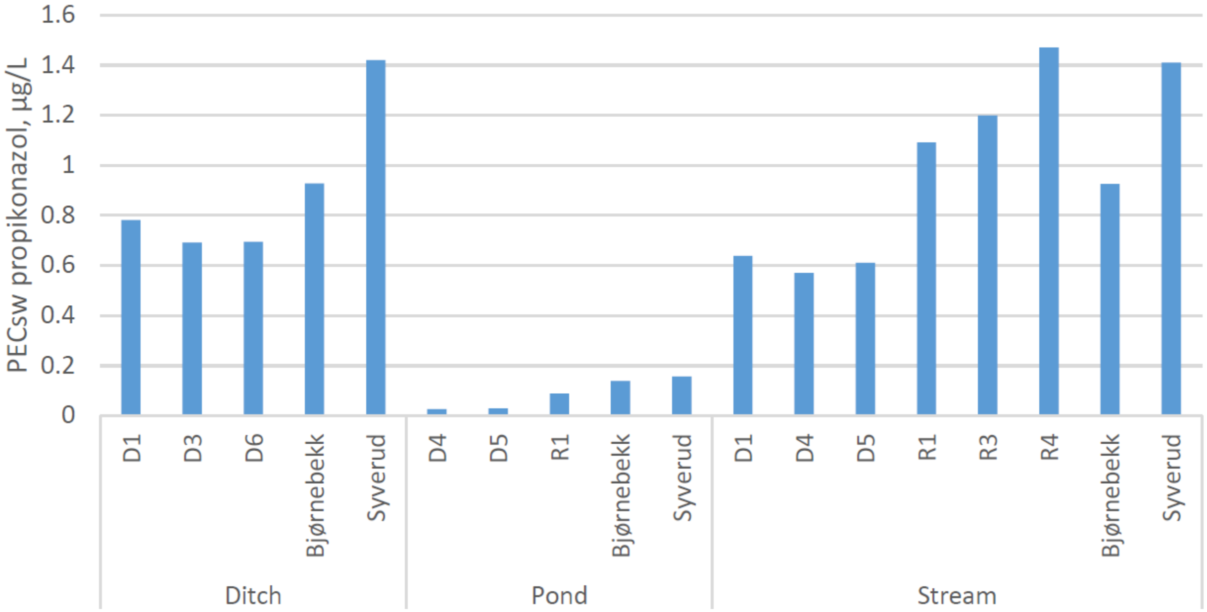


Figure 5.4.2-3 Predicted concentration of propiconazole in ditch, pond and streams using FOCUS Swash scenarios and the Norwegian scenarios using WISPE.

From this exercise, the European scenarios seems to cover the conditions in Norway and trusting the model simulations from Syverud and Bjørnebekk these scenarios give good protection for the selected soil type and climate. For condition along the west-coast with more precipitation and areas with slower degradation the picture could be different.

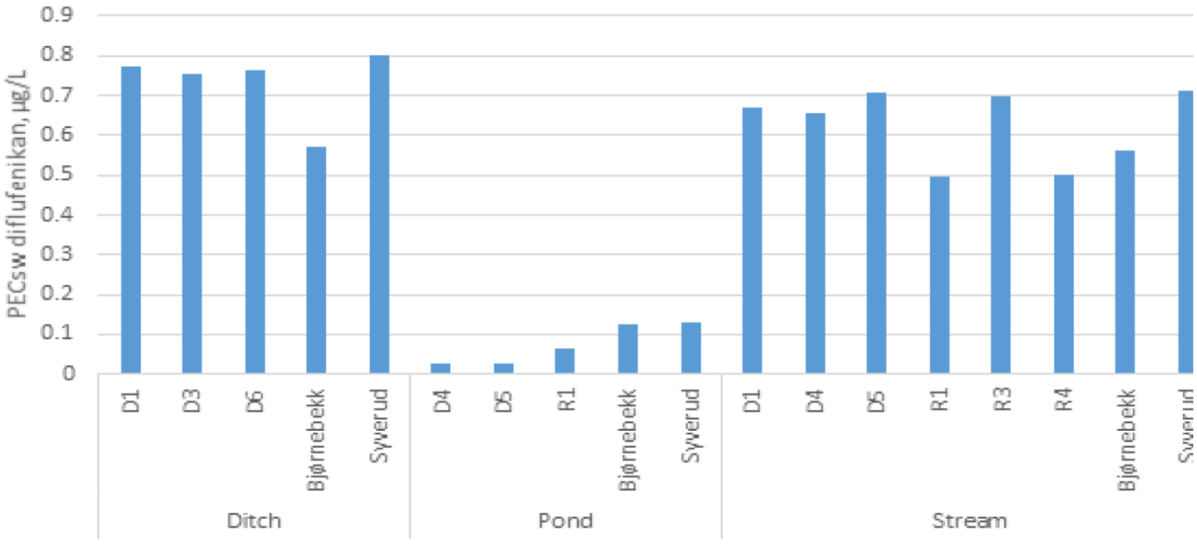


Figure 5.4.2-4 Predicted concentrations of diflufenican in ditch, pond and streams using the FOCUS Swash scenarios and the Norwegian scenarios using WISPE.

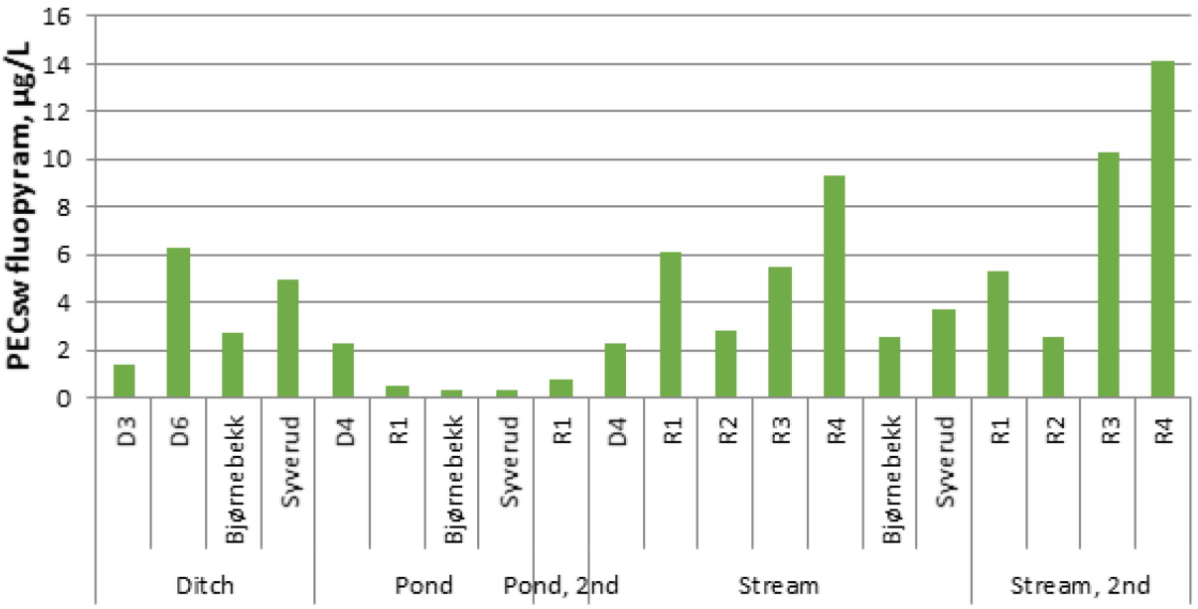


Figure 5.4.2-5 Predicted concentrations of fluopyram in ditch, pond and streams using the FOCUS swash scenarios and the Norwegian scenarios using WISPE.

To look at the effect of using a national endpoint of half-life, the Surface water scenario from Syverud was used with the "European" DT 50 = 119 days (Figure 5.4.2-6) and the "Norwegian" DT50 = 900 days (Figure 5.4.2-7).

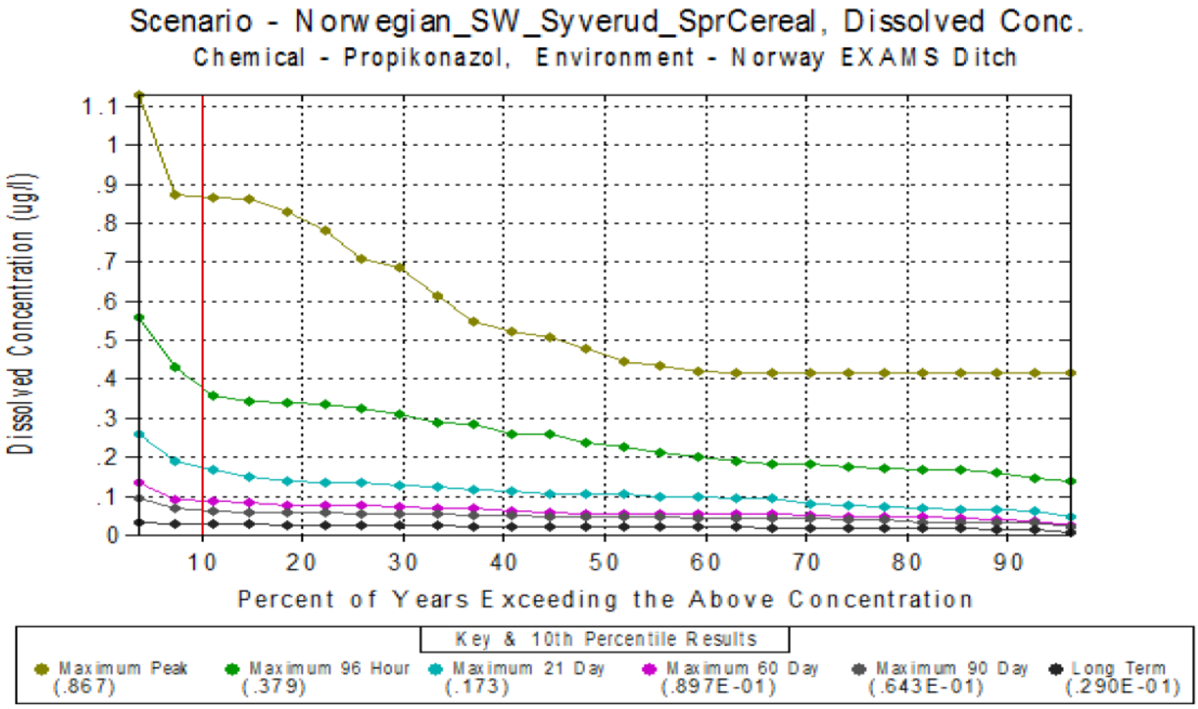


Figure 5.4.2-6 Simulated concentration of propiconazole in surface runoff from cereals, using Syverud scenario (20 years) with WISPE and DT50=119 days.

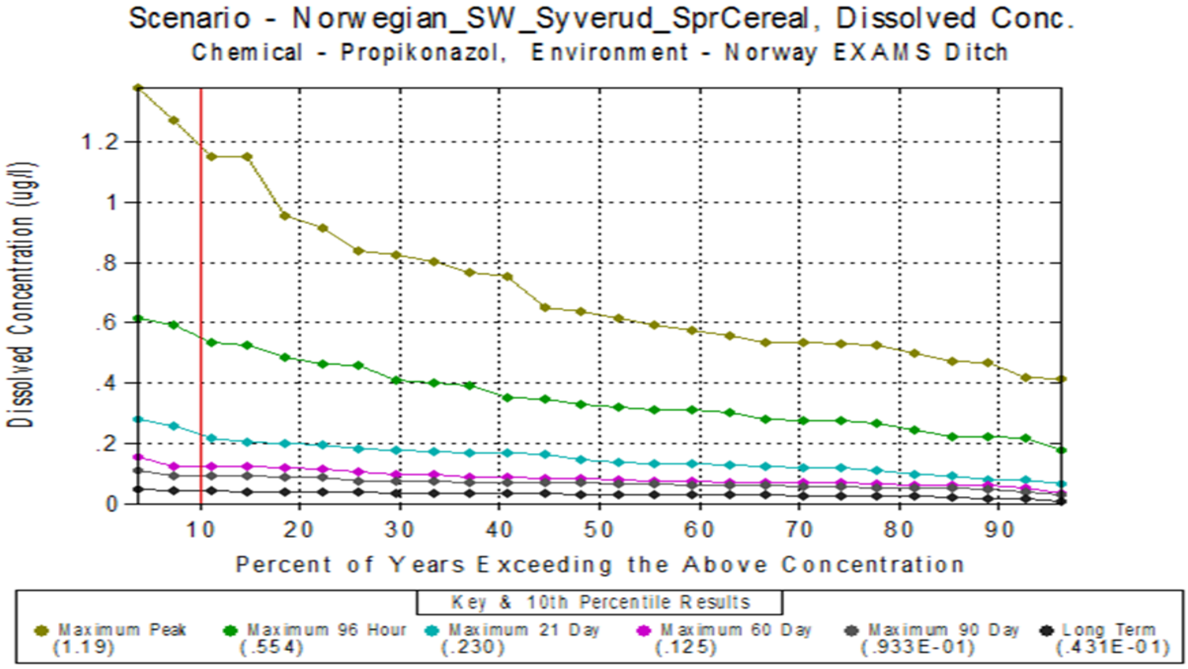


Figure 5.4.2-7 Simulated concentration of propiconazole in surface runoff from cereals, using Syverud scenario (20 years) with WISPE and DT50=900 days.

Simulations with different degradation rate seem not to influence the concentration of propiconazole in the runoff. The reason is probably that transport is mainly controlled by sorption. However, the choice of DT50 has a significant effect on the mass-balance of propiconazole in the soil as shown in Figure 5.4.2-8.

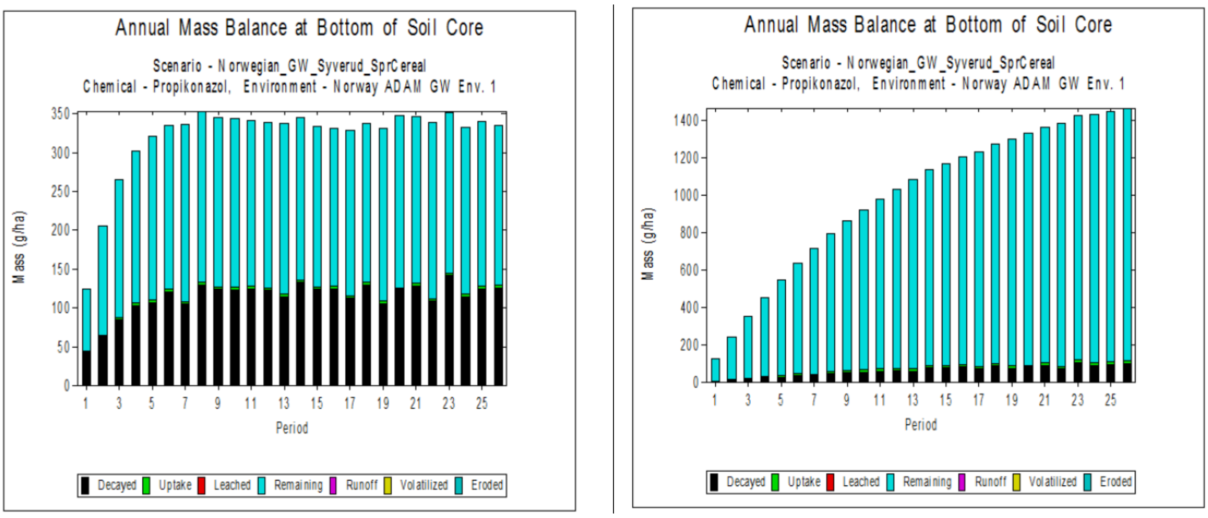


Figure 5.4.2-8 Annual mass balance at bottom of the soil core after 26 year of propiconazole application, 125g/ha. DT50 =119 day (left) and 900 days (right).

5.4.3 Conclusion surface water

From the simulation exercise with WISPE and FOCUS with the FOCUS parameters, there is small differences between Norwegian scenarios with WISPE and the FOCUS scenarios including mobile and more strongly sorbed pesticides. Changing input parameters with site-specific degradation endpoint the situation is expected to change, but experience from using national endpoint for Norway in surface water modelling is lacking. From a preliminary exercise with strongly sorbed pesticides seems not to change concentration in surface water significantly, but concentrations in sediment and soil core are accumulating.

5.5 Conclusions from the scenario modelling

The existing Norwegian scenarios in groundwater and surface water seem to be representative in the meaning of covering the main soil types in the central agricultural areas in South Eastern Norway. However there are no scenarios covering areas of South Western and Northern Norway characterized by soil with high organic content, slow degradation and heavy rainfall. Certain, smaller agricultural areas in Norway are considered to be more prone to leaching and/or runoff than the areas covered by the two Norwegian scenarios. Such, vulnerable areas are those with high groundwater levels and sandy soil, with high risk of leaching of mobile pesticides, and hilly areas with clay soil, which represent high risk of surface runoff with strongly sorbed pesticides. These situations have to be considered separately in the risk assessments.

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