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Are spray drift losses to agricultural roads more important for surface water contamination than direct drift to surface waters?

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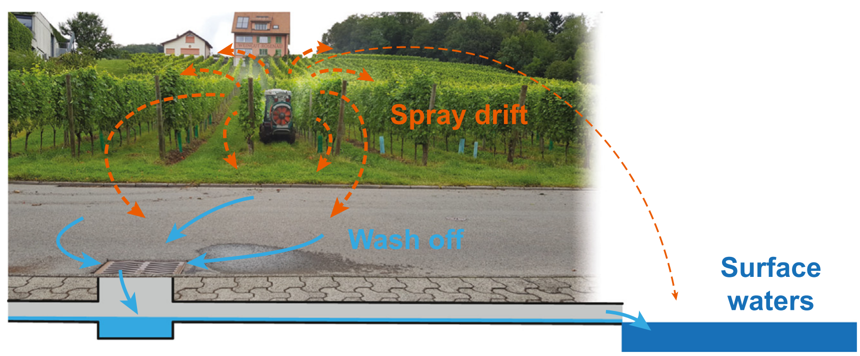
HIGHLIGHTS

- We evaluated the potential of spray drift wash-off from roads to surface waters.
- Analysis of 26 Swiss catchments: Field mapping combined with spatial modelling.
- Drift to roads draining to surface waters is much larger than drift to surface waters.
- Major fractions of the drift deposited on roads can be washed off during rainfall.
- High risk in vineyards and for pesticides with low adsorption coefficients.

GRAPHICAL ABSTRACT



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ABSTRACT

Spray drift is considered a major pesticide transport pathway to surface waters. Current research and legislation usually only considers direct spray drift. However, also spray drift on roads and subsequent wash-off to surface waters was identified as a possible transport pathway. Hydraulic shortcuts (storm drainage inlets, channel drains, ditches) have been shown to connect roads to surface waters, thus increasing the risk of drift wash-off to surface waters. However, the importance of this pathway has never been assessed on larger scales. To address this knowledge gap, we studied 26 agricultural catchments with a predominance of arable cropping ($n = 17$) and vineyards ($n = 9$). In these study sites, we assessed the occurrence of shortcuts by field mapping. Afterwards, we modelled the areas of roads drained to surface waters using a high-resolution digital elevation model (0.5 m resolution) and a multiple flow algorithm. Finally, we modelled drift deposition to drained roads and surface waters using a spatially explicit, georeferenced spray drift model. Our results show that for most sites, the drift to drained roads is much larger than the direct drift to surface waters. In arable land sites, drift to roads exceeds the direct drift by a factor of 4.5 to 18, and in vineyard sites by 35 to 140. In arable land sites, drift to drained roads is rather small (0.0015% to 0.0049% of applied amount) compared to typical total pesticide losses to surface waters. However, substantial drift to drained roads in vineyard sites was found (0.063% to 0.20% of applied amount). Current literature suggests that major fractions of the drift deposited on roads can be washed off during rain events, especially for pesticides with low soil adsorption coefficients. For such pesticides and particularly in vineyards, spray drift wash-off from drained roads is therefore expected to be a major transport pathway to surface waters.

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1. Introduction

After pesticide application on agricultural crops, a certain fraction of the applied amount is not attained to the target crop, but is lost to non-target ecosystems such as surface waters. These pesticide losses pose a major threat for aquatic ecosystems (Schwarzenbach et al., 2006; Stehle and Schulz, 2015). Besides point sources (e.g. farmyard runoff, accidental spills, combined sewer overflows, or and wastewater treatment plants), surface runoff (Larsbo et al., 2016; Reichenberger et al., 2007), macropore flow to tile drainages (Kladivko et al., 2001; Sandin et al., 2018), and spray drift (Carlsen et al., 2006; Ganzelmeier, 1995) are considered the most important pesticide transport pathways to surface waters. Spray drift is defined as the downwind movement of spray droplets beyond the target area of application originating from the spraying process (Stephenson et al., 2006). Studies quantifying surface water pollution by spray drift are typically only considering drift directly deposited on surface waters. This holds for modelling studies (Huber et al., 2000; Padovani et al., 2004; Röpke et al., 2004; Travis and Hendley, 2001), field studies (Bonzini et al., 2006; Schulz, 2001), and is also the case for the models used in the European pesticide authorisation (Linders et al., 2003). However, spray drift is also deposited on various other non-target areas (e.g. soils, non-target crops, forests, settlements, roads, farm tracks). Depending on the spraying device, the non-target deposition is estimated to 0.8 – 4% of the applied amount for ground applications (Jensen and Olesen, 2014; Viret et al., 2003). Depending on the properties of the non-target area, some of this spray drift may be washed off to surface waters during subsequent rainfall events (Gassmann et al., 2013; Schönenberger et al., in review).

Roads and farm tracks have a very low infiltration capacity and limited sorption potential (Ramwell, 2005). Therefore, on these areas, surface runoff is formed with higher frequency and pesticides are washed off much easier than from target areas or from other non-target area types. Especially substances with low soil adsorption coefficients (K_{oc} < 250 mL g^{-1}) have been shown to be washed off in large amounts (57% or more of the applied amount) during simulated and natural rainfall (Ramwell et al., 2002; Thuyet et al., 2012). However, also for substances with higher K_{oc}, relevant wash-off fractions have been reported during the first rainfall after application, e.g. up to 5.8% (Thuyet et al., 2012) and up to 2.7% (Jiang et al., 2012).

Roads and farm tracks in agricultural areas are often drained by storm water drainage inlets or by other artificial structures (e.g. channel drains or ditches) (Alder et al., 2015; Payraudeau et al., 2009; Rübel, 1999; Schönenberger and Stamm, 2021). Especially in Switzerland, these structures are often connected to surface waters via subsurface pipe systems. This enables spray drift wash-off from remote roads to reach surface waters and therefore creates a so-called shortcut (Doppler et al., 2012). These shortcuts therefore strongly increase the potential of spray drift wash-off from roads for surface water pollution.

Despite its large potential for pesticide transport to surface waters, only in four catchments measurements providing insights on this transport process were performed to the best of our knowledge: In a German vineyard catchment, Rübel (1999) found that drift on vineyard roads during helicopter applications was leading to high pesticide concentrations in the receiving stream in the following rain event. Ground applications were found to have a similar effect, but were leading to lower maximal concentrations compared to helicopter spraying. In a French vineyard catchment, Lefrancq et al. (2014) reported spray drift on roads and subsequent wash-off to be responsible for a large fraction of the runoff-related fungicide load at the catchment outlet. In a Swiss arable land catchment, Schönenberger et al. (in review) found that either spray drift on roads or spills from leaking spraying equipment led to increased pesticide concentrations in inlets of the road storm water drainage system. Finally, in another Swiss arable land catchment, Ammann et al. (2020) found – based on the field study described in Doppler et al. (2012) – that the consideration of spray drift wash-off from roads could strongly reduce the uncertainty of exposure models.

These studies show that spray drift wash-off from roads is a relevant transport pathway to surface waters in certain catchments. However, it remains unclear how much spray drift is deposited on roads draining to surface waters for larger spatial scales, and how the amount deposited differs between catchments and crop types. In addition, it is unknown to which degree drift reduction measures (e.g. spray drift buffers) could reduce pesticide losses caused by this pathway.

For assessing spray drift to surface waters (usually streams, but also ditches or ponds) on larger spatial scales, various studies have applied spatially explicit georeferenced drift models (Holterman and Van de Zande, 2008; Kubiak et al., 2014; Schad and Schulz, 2011; Wang and Rautmann, 2008). These models combine spatial data on surface waters and sprayed crops with spray drift deposition functions obtained from experimental trials (Ganzelmeier, 1995; Rautmann et al., 1999). Similar models have been used for assessing spray drift to other non-target areas, such as terrestrial habitats (de Jong et al., 2008). However, to our knowledge, such models have not been applied to roads or farm tracks.

In this study, we therefore aimed at comparing spray drift deposition on surface waters to the deposition on roads and farm tracks draining to surface waters. For this, we combined a field mapping approach with a spatially explicit, georeferenced spray drift model for a large set of agricultural catchments representing arable land and vineyards in Switzerland. We focused on these two crop types since they are two of the most important crop types in Switzerland with respect to coverage (arable land) and average pesticide use (vineyards). Additionally, spray drift deposition differs strongly between those two crop types due to different spraying methods (boom sprayers on arable land, and air blast sprayers on vineyards), and different spatial structures (e.g. density and size of roads around crop areas) (Schönenberger and Stamm, 2021). In some Swiss vineyard regions, also helicopters are still used for spraying. This method is however not addressed in this study.

Our research questions are:

- How much spray drift is deposited on roads and farm tracks draining to surface waters? In comparison, how much spray drift is deposited in surface waters directly?
- How do the deposited amounts differ between arable land and vineyards?
- How much could the drift on drained roads and farm tracks be reduced by spray drift buffers?

Based on the respective results, we also aim at answering the question how important spray drift wash-off from roads may be for the pesticide pollution of surface waters compared to direct spray drift, and compared to total pesticide losses. However, given the paucity of empirical data on wash-off from these surfaces the results will be only tentative at this stage.

2. Methods

2.1. Selection of study sites

We selected two sets of agricultural catchments as study sites for our analysis. One set represents Swiss arable land areas, the other one Swiss vineyards. The arable land and vineyard sites were selected randomly from a nationwide, small-scale topographic catchment dataset (FOEN, 2012). The selection probability of each catchment equalled the arable land and vineyard area in the catchment, respectively, as reported by the Swiss land use statistics (FSO, 2014) (details – Schönenberger and Stamm, 2021 for arable land, Simon (2019) for vineyards). From the resulting sites (20 arable land, 8 vineyards), we removed four arable land sites for which no high-resolution crop data were available. Additionally, two vineyard sites only consisted of small-scale plots in settlements. Since this type of small-scale viticulture is a special case present

Table 1
Overview over study sites. Selection: R – random selection, M – site used in previous studies.

ID	Study site	Canton	Abbreviation	Crop type	Selection	Area (km^2)
1	Böttstein	AG	BOETT	Arable	R	3.34
2	Boncourt	JU	BONCO	Arable	R	5.90
3	Buchs	ZH	BUCHS	Arable	R	3.86
4	Clarmont	VD	CLARM	Arable	R	2.47
5	Courroux	JU	COURR	Arable	R	2.80
6	Hochdorf	LU	HOCHD	Arable	R	2.37
7	Illighausen	TG	ILLIG	Arable	R	1.90
8	Molondin	VD	MOLON	Arable	R	4.15
9	Müswangen	LU	MUESW	Arable	R	3.00
10	Nürensdorf	ZH	NUERE	Arable	R	2.34
11	Oberneunforn	TG	OBERN	Arable	R	3.30
12	Schalunen	BE	SCHAL	Arable	M	2.78
13	Suchy	VD	SUCHY	Arable	R	3.28
14	Truttikon	ZH	TRUTT	Arable	R	5.06
15	Ueken	AG	UEKEN	Arable	R	1.99
16	Vufflens-la-Ville	VD	VUFFL	Arable	R	2.79
17	Meyrin (arable)	GE	MEY-A	Arable	R	8.50
18	Bex	VD	BEX	Vineyard	R	4.27
19	Bourg-en-Lavaux	VD	BOURG	Vineyard	R	0.67
20	Cornaux	NE	CORNA	Vineyard	R	2.76
21	Fläsch	GR	FLAES	Vineyard	R	2.29
22	Hallau	SH	HALLA	Vineyard	M	0.98
23	Meyrin (vineyard)	GE	MEY-V	Vineyard	R	1.50
24	Mont-Vully	FR	MONTV	Vineyard	R	1.63
25	Savièse	VS	SAVIE	Vineyard	M	2.41
26	Saxon	VS	SAXON	Vineyard	R	4.25
				Average		3.22

only in few areas, these two sites were also removed. In contrast to the other sites, the site Meyrin contains both, large arable land areas and large vineyards. This site was therefore splitted into an arable land part and a vineyard part (see Table 1). Finally, the selected sites were complemented by three catchments used in previous studies assessing pesticide concentrations in surface waters (Schönenberger et al., in review; Spycher et al., 2018; Spycher et al., 2019). The resulting 26 sites (17 arable land, 9 vineyards) are shown in Fig. 1.

2.2. Modelling procedure

For this study, we considered two types of non-target areas: Surface waters and drained roads. Other non-target areas (e.g. hedges, plot margins) were considered irrelevant for subsequent transfer to surface waters since surface runoff formation on these areas is rare compared to roads (see Section 1). Drained roads were defined as roads from which water drains to surface waters while only flowing along roads or through shortcuts. They were categorized into roads draining to surface waters via shortcuts and into roads directly draining to surface waters. For determining drained roads, we first mapped shortcuts in the study catchments and then combined these maps with a flow path model (see Fig. 2). Afterwards, we determined the amount of spray drift deposited on drained roads and on surface waters using a spray drift model. In the following, these steps are described in detail.

2.2.1. Mapping shortcuts

Shortcuts were defined as artificial structures increasing and/or accelerating the process of surface runoff reaching surface waters (Schönenberger and Stamm, 2021). Within this study, we considered storm drainage inlet shafts, channel drains, and ditches along roads and farm tracks as potential shortcuts. These potential shortcuts were defined as real shortcuts, if they are drained to surface waters, to wastewater treatment plants or to combined sewer overflows.

In all 26 study sites, we mapped potential shortcuts along roads and farm tracks. For the arable land sites, mapping was performed in 2017 and 2018 for the whole catchments, combining three different methods: Field surveys, storm drainage system plans, and high resolution aerial images (resolution: 2.5 to 5 cm) from an unoccupied aerial vehicle (details – Schönenberger and Stamm, 2021). For vineyard sites, mapping

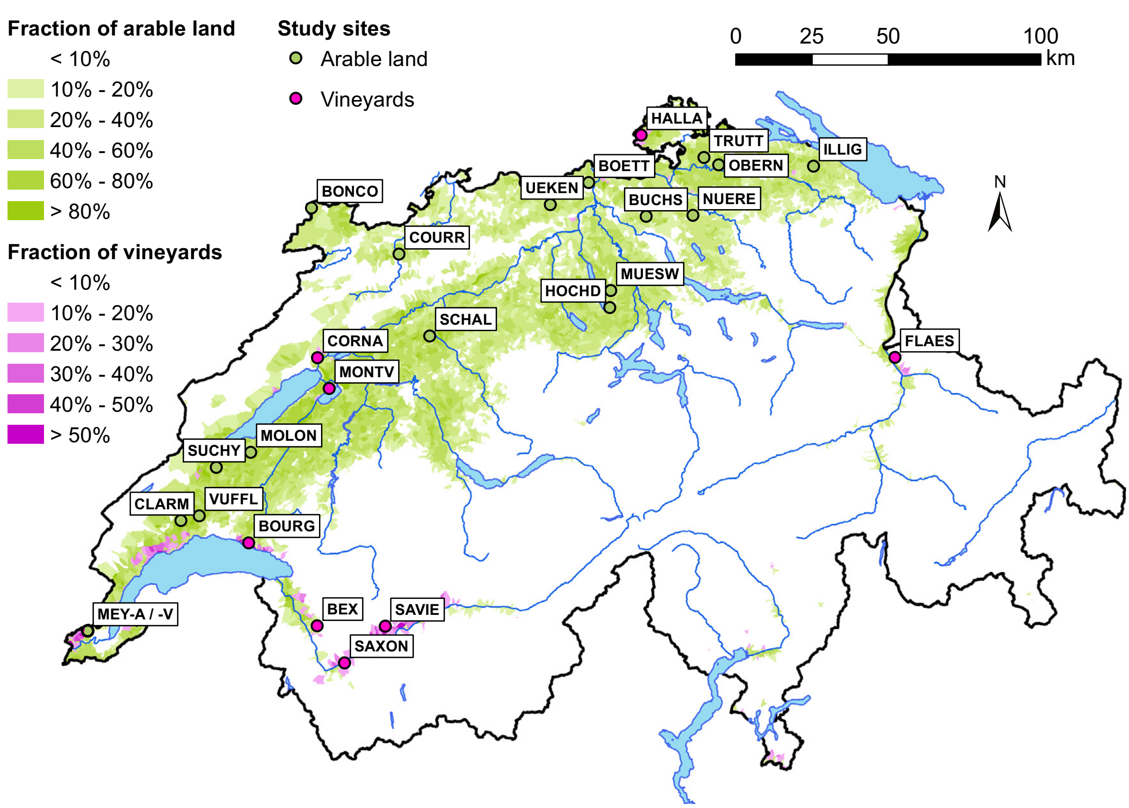

Fig. 1. Locations of the study sites and fractions of arable land and vineyards in Swiss hydrological catchments. Sources: FOEN (2012), FSO (2014), Swisstopo (2010).

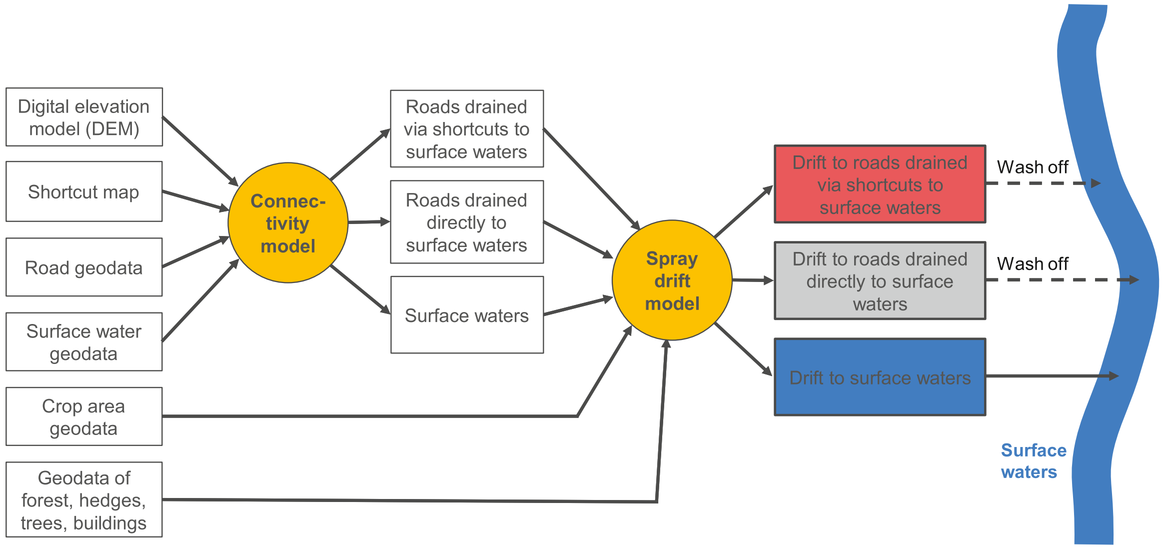

Fig. 2. Schematic representation of the procedure for modelling spray drift to roads drained via inlets, to roads drained directly to surface waters, and to surface waters.

was performed in 2019. In contrast to arable land, which is often distributed throughout the whole catchment, vineyards usually only cover a certain part of the catchment. We therefore did not map potential shortcuts in the whole catchment, but only along roads and farm tracks adjacent to vineyards. The mapping in vineyards was based on field surveys (details – Simon (2019)), complemented with data from storm drainage system plans, Google Street View, and aerial images with intermediate resolution (10 cm) (Swisstopo, 2019).

If storm drainage plans were available for the respective study site, we additionally determined where potential shortcut structures drain to. Structures draining to surface waters, wastewater treatment plants and/or combined sewer overflows were considered as real shortcuts. Structures draining to infiltration areas (e.g. infiltration ponds, forests, or grassland) were not considered as real shortcuts and were neglected in the further steps. Ninety-nine percent of the storm drainage inlets, and 98% of the channel drains and ditches were found to be real shortcuts in a previous study (Schönenberger and Stamm, 2021). Therefore, potential shortcuts for which no drainage plans were available were assumed to act as real shortcuts.

2.2.2. Surface runoff connectivity model

To determine drained roads in the study sites, we used a modified version of the surface runoff connectivity model described in Schönenberger and Stamm, 2021. This was done in four steps as described below. How the required model parameters were chosen and how their influence on the model results was assessed, is described in Section 2.2.4.

1) *Determination of road areas.* Road and farm track areas (called road areas in the following) were derived from the topographical landscape model swissTLM3D (Swisstopo, 2020b), and were complemented with other sealed areas from the same dataset (parking lots, motorway stations). Details are given in the appendix (A1.1).
2) *Determination of surface water areas.* Surface water areas (streams and stagnant waters) were also derived from the topographical landscape model swissTLM3D (Swisstopo, 2020b). Details are given in the supporting information (A1.2).
3) *Determination shortcut areas.* Shortcut areas were defined as the mapped shortcut structures and the area extending 1 m from these structures.
4) *Determination of connectivity.* For determining the connectivity of road areas to surface water or shortcut areas, we used the TauDEM model (Tarboton, 1997), which is based on a D-infinity flow direction algorithm. As an input, we used a digital elevation model (DEM) with a resolution of 0.5 m (Swisstopo, 2020a) that was modified as follows. Firstly, to account for the surface runoff accumulation effect of roads (Dehotin et al., 2015; Fiener et al., 2011; Heathwaite et al., 2005), road areas were carved into the DEM by a certain depth (parameter *road carving depth* d_{road}). The surface runoff accumulation effect describes the characteristic of roads to concentrate runoff along their course, mainly by acting as a barrier for diffuse surface runoff from adjacent fields. Since the accuracy of DEM is often not high enough to reproduce this effect, it is created artificially by carving roads into the DEM. Secondly, all topographic sinks smaller than a certain depth (parameter *sink fill depth* d_{sink}) were filled. Finally, surface water areas and shortcut areas were carved 50 m and 20 m into the DEM. These large carving depths ensured that raster cells representing surface water and shortcut areas were much lower than the surrounding terrain. This guaranteed that the flow direction of the raster cells adjacent to surface water and shortcut areas pointed towards these areas. The modified DEM, shortcut areas, and surface water areas were then used as an input for the D-infinity upslope dependence tool of the TauDEM model. As a result, we obtained a raster containing all roads drained to surface waters or shortcuts. Some of the raster cells classified as drained roads had a flow path running for longer distances over fields or through forests. However, we expect runoff formed on roads to infiltrate when flowing for longer distances on these areas. Therefore, we removed drained road cells from the raster dataset if their flow path outside roads exceeded a maximal distance (parameter *infiltration distance* d_{inf}).

To assess which area of drained roads per crop area is found per study site s and how this compares to the area of surface waters, we calculated drainage densities d_s (drained area per crop area) as follows:

$$d_s = \begin{pmatrix} d_{RSC,s} \\ d_{RSW,s} \\ d_{SW,s} \end{pmatrix} = \frac{\begin{pmatrix} A_{RSC,s} \\ A_{RSW,s} \\ A_{SW,s} \end{pmatrix}}{A_{crop,s}} \quad (1)$$

$d_{RSC,s}$, $d_{RSW,s}$, and $d_{SW,s}$ are the drainage density of roads drained to shortcuts, the drainage density of roads drained to surface waters, and the drainage density of surface waters in study site s. $A_{RSC,s}$, $A_{RSW,s}$, and $A_{SW,s}$ are the areas of roads drained to shortcuts, of roads drained to surface waters, and of surface waters in study site s. $A_{crop,s}$ is the crop area in study site s.

2.2.3. Spray drift model

The spray drift model developed in this study determines drift from crop areas to the relevant non-target areas (i.e. drained roads and surface waters) based on their spatial arrangement in the study sites. Additionally, the model considers drift reduction by barriers, such as forest, hedges, trees, or buildings. In this section, we first describe how the input data were prepared, and afterwards how spray drift was modelled. In Section 2.2.4, we describe how model parameters were chosen and how the model uncertainty was assessed.

2.2.3.1. Input data. As input data for the spray drift model, we used the areas of drained roads and of surface waters, determined by the connectivity model (Section 2.2.2). Drained roads and streams were rasterized with a resolution of 2x2m. Larger surface waters (e.g. ponds, lakes, large rivers) were rasterized with a resolution of 10 × 10 m. The areas of forest, hedges, trees, buildings, and vineyards were obtained from the topographical landscape model swissTLM3D (Swisstopo, 2020b). This dataset does however not specify the extent of arable land. Arable land areas were therefore extracted from a collection of standardized cantonal datasets on agricultural areas in parcel resolution (Kanton Aargau et al., 2020).

We assumed that pesticides are applied according to Swiss regulations, Swiss proof of ecological performance (ChemRRV, 2005; DZV, 2013), and good agricultural practice. These regulations prohibit pesticide applications within a buffer of 6 m around surface waters, 3 m around hedges, forests, and riparian woods, and 0.5 m around roads and farm tracks. For our analysis, we therefore removed all crop areas (vineyards, arable land) lying inside these buffers.

2.2.3.2. Spray drift model. The spray drift model developed in this study is based on spray drift curves according to Rautmann et al. (1999). They describe the spray drift deposition $\rho_{drift,i,p}$ (kg m^{-2}) on a non-target area i depending on its upwind distance $d_{i,p}$ (m) to a sprayed plot p (Eq. (2)), and on the crop-specific spray drift parameters a and b. $\rho_{appl,p}$ is the application rate (kg m^{-2}) on the sprayed plot.

$$\rho_{drift,i,p} = a \cdot {d_{i,p}}^{b} \cdot \rho_{appl,p} \tag{2}$$

The spray drift parameters a and b were derived in field trials for wind speeds between 1 and 5 m/s and are therefore only valid for these wind speeds. Higher wind speeds would lead to an increase in drift deposition. However, according to good agricultural practice, farmers should not apply pesticides for wind speeds higher than 5 m/s. The minimal drift distances measured in the field trials were 1 m for arable land and 3 m for vineyards. The maximal distances were 100 m for both trial types. For this study, we extrapolated the drift curve to a minimal drift distance of 0.5 m, and to a maximal distance as defined by the parameter *maximal drift distance* $d_{drift,max}$. For distances larger than this parameter, the spray drift deposition was set to zero.

The upwind distance from a non-target area cell to the next sprayed plot depends strongly on the wind direction. Similar to other studies (e.g. Wang and Rautmann (2008), Golla et al. (2011)), we therefore calculated the upwind distances for eight different wind directions (N, NE, E, SE, S, SW, W, NW). In the field trials used for the determination of spray drift curves, sprayed plots had a standardized width of around 20 m parallel to the wind direction (Julius Kühn-Institut, 2013). However, in the study sites analysed here, the extent of crop areas along the wind line was often larger than these 20 m. In these cases, we assumed the drift of these crop areas to equal the drift produced by a sequence of standard plots located in intervals of 20 m along the wind line (example – Fig. 3). For each of these standard plots, drift was calculated separately, summed up, and multiplied with the area A_i (m^2) of

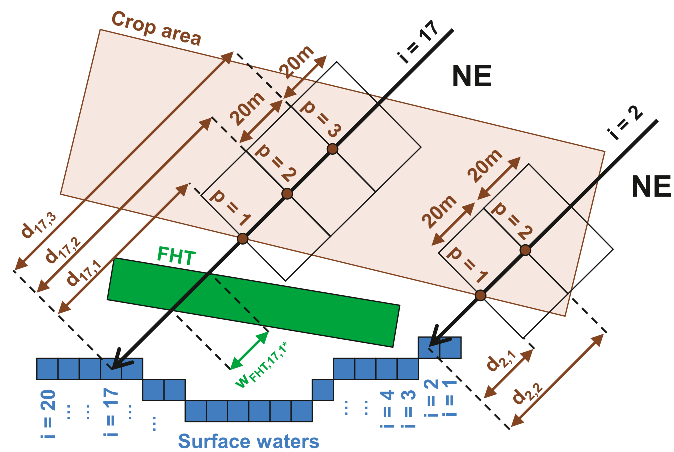

Fig. 3. Example of the calculation of drift distances $d_{i,p}$ and barrier widths $w_{FHT,i,p}$ for two non-target area cells ($i = 2$, and $i = 17$) for the wind direction northeast (NE). In this example, the non-target area cells are surface waters. Forest, hedges and trees (FHT), but no buildings act as a barrier. *The barrier widths $w_{FHT,17,2}$ and $w_{FHT,17,3}$ are in this case equal to the barrier width $w_{FHT,17,1}$.

the non-target area cell to determine the spray drift $m_{drift,i,w}$ (kg) for wind direction w to the non-target area cell i (Eq. (3)).

$$m_{drift,i,w} = \sum_{p=1}^{n_p} \left(a \cdot {d_{i,p,w}}^{b} \cdot \rho_{appl,p} \right) \cdot A_i \tag{3}$$

In the model, we additionally considered the interception of spray drift by barriers, such as forests, hedges, trees, or buildings. For forest, hedges, and trees (FHT), we assumed that drift is linearly reduced proportional to their width $w_{FHT,i,p,w}$ (m) along the wind line (see Fig. 3) between the sprayed plot and the non-target area. The amount of drift reduction is described by the drift reduction factor $f_{FHT,i,p,w}$ (Eq. (4)). The distance needed for intercepting all spray drift is described by the model parameter *width of forest, hedges, or trees causing full drift interception* $w_{FHT,int}$ (m). An example of how $w_{FHT,i,p,w}$ is calculated if an FHT polygon is located further away from the non-target area is provided in Section A1.3.

$$f_{FHT,i,p,w} = \max\left(1 - \frac{w_{FHT,i,p,w}}{w_{FHT,int}}, 0\right) \tag{4}$$

Similarly, a drift reduction factor for buildings $f_{B,i,p,w}$ was added to the model (Eq. (5)). If a building is located between the sprayed plot and the non-target area, the spray drift is reduced as specified by the model parameter *spray drift interception by buildings* $f_{B,int}$. $w_{B,i,p,w}$ is the width of buildings between the sprayed plot and the non-target area along the wind line.

$$f_{B,i,p,w} = \begin{cases} 1 | w_{B,i,p,w} = 0 \\ 1 - f_{B,int} | w_{B,i,p,w} > 0 \end{cases} \tag{5}$$

In a last step, we combined Eqs. (3) to (5). Assuming that the application rate ρ_{appl} is the same for all crop areas per study site, the amount of spray drift lost per total amount applied per study site $f_{lost,w}$ was calculated as shown in Eq. (6). For each study site, the amount of spray drift lost per applied amount was calculated for all three non-target area types, and all eight wind directions. Additionally, we calculated the relative losses $f_{lost,rel}$ to each non-target area type (Eq. (7); RSC – roads drained to shortcuts, RSW – roads drained to surface waters, SW – surface waters).

Table 2
Model parameters used as reference parameter set, for the sensitivity analysis, and for extreme estimates: Reference parameter set (p_{ref}), parameters used for the sensitivity analysis (p_{sens}), parameter sets for minimal and maximal total drift (p_{min}, p_{max}), parameter sets for minimal and maximal relative drift to surface waters ($p_{SWrel,min}$, $p_{SWrel,max}$). Model results were not sensitive to changes of parameters marked with a star (*) (see Section 3.3). Therefore, these parameters were kept constant when assessing the maximal and minimal drift.

Model	Parameter	p_{ref}	p_{sens}	p_{min}	p_{max}	$p_{SWrel,min}$	$p_{SWrel,max}$
Connectivity	Road carving depth d_{road}	10 cm	0 cm, 5 cm, 10 cm, 15 cm, 20 cm	0 cm	20 cm	20 cm	0 cm
Connectivity	Sink fill depth d_{sink}	20 cm	10 cm, 20 cm, 50 cm	20 cm*	20 cm*	20 cm*	20 cm*
Connectivity	Infiltration distance d_{inf}	20 m	5 m, 10 m, 20 m, 30 m	10 m	30 m	30 m	10 m
Spray drift	Maximal drift distance $d_{drift,max}$	100 m	100 m, 175 m, 250 m	100 m	250 m	100 m	250 m
Spray drift	Width of forest, hedges, and trees causing full drift interception $w_{FHT,int}$	10 m	5 m, 10 m, 15 m, 20 m	5 m	20 m	5 m	20 m
Spray drift	Drift interception by buildings $f_{B,int}$	100%	0%, 100%	100%*	100%*	100%*	100%*

$$f_{lost,w} = \frac{\sum_{i=1}^{n_i} m_{drift,i,w}}{m_{appl}} = \frac{\sum_{i=1}^{n_i}\left(\sum_{p=1}^{n_p}\left(a \cdot d_{i,p,w}{}^{b} \cdot f_{FHT,i,p,w} \cdot f_{B,i,p,w}\right) \cdot A_i\right) \cdot \rho_{appl}}{A_{crop} \cdot \rho_{appl}} \quad (6)$$

$$f_{lost,rel} = \begin{pmatrix} f_{lost,rel,RSC} \\ f_{lost,rel,RSW} \\ f_{lost,rel,SW} \end{pmatrix} = \frac{\begin{pmatrix} f_{lost,RSC} \\ f_{lost,RSW} \\ f_{lost,SW} \end{pmatrix}}{f_{lost,RSC} + f_{lost,RSW} + f_{lost,SW}} \quad (7)$$

For the crop-specific drift parameters a and b, we used an updated version of the median spray drift parameters of Rautmann et al. (1999) provided by the authors of the publication. For arable land, they equalled 0.9658 (a) and − 0.9507 (b), for vineyards 30.408 (a) and − 1.5987 (b).

2.2.4. Parameter range and sensitivity analysis

The connectivity model and the spray drift model have three model parameters each. For all parameters we selected a parameter range based on field experience or literature. As a reference parameter value, we additionally selected a single value within this range that seemed the most realistic to us.

The ranges of the connectivity model parameters (road carving depth d_{roads}, sink fill depth d_{sink}, and infiltration distance d_{inf}) were chosen based on our field experience from shortcut mapping and on our prior knowledge on surface runoff along roads (see Table 2). For the road carving depth $d_{roads,}$ we included the value 0 m (i.e. no change to the elevation model) as the lower end of the parameter range. However, we do not think that this value is able to represent the surface runoff accumulation effect of roads properly. To validate the results of the connectivity model, a flow path map of the study site Schalunen (Schönenberger et al., in review) was qualitatively compared to the model results.

In contrast to the connectivity model parameters, the spray drift model parameters were chosen based on literature values. The spray drift curves of Rautmann et al. (1999) were obtained by measuring distances up to 100 m from the sprayed plot. Therefore, for the parameter max*imal drift distance* $d_{drift,max}$, 100 m was chosen as the parameter range minimum, and as reference parameter. Since Rautmann et al. (1999) state that the curve can also be extrapolated up to 250 m, we set this distance as the parameter range maximum.

Various studies have assessed the drift intercepting properties of hedges (also known as windbreaks). For example, Wenneker and van de Zande (2008) found a reduction of 80–90% for hedges with a width of 1 to 1.25 m in full leaf stage. Other studies report a reduction between 68% to more than 90% depending on leaf density and wind speed (Ucar and Hall, 2001). These studies show that the width of forest, hedges, or trees causing a full spray drift reduction varies depending on various factors (e.g. leaf density, height, vegetation period) and can therefore not be quantified by a single value. The model parameter *width to full drift interception of forest, hedges, and trees* $w_{FHT,max}$ was therefore varied within a realistic range, based on the available data (i.e. between 5 and 20 m).

In contrast to forest, hedges, and trees, we expect buildings to completely intercept spray drift. Therefore, we set the reference parameter *drift interception by buildings* $f_{B,int}$ to 100%. However, in the sensitivity analysis, we also tested the effect of completely ignoring this process ($f_{B,int} = 0\%$).

To assess the influence of model parameters to our results, we performed a local sensitivity analysis starting from the reference parameter set and varying each parameter separately. Additionally, based on the results of the local sensitivity analysis, we combined the parameters such that they lead to extreme estimates, i.e. minimal and maximal estimate of total drift to non-target areas f_{lost} (p_{min}, p_{max}), and minimal and maximal estimate of drift deposited on surface waters relative to the drift lost on surface waters and drained roads $f_{lost,rel,SW}$ ($p_{SWrel,min}$, $p_{SWrel,max}$; see Table 2).

3. Results and discussion

3.1. Drainage densities

As a result of the surface runoff connectivity model, we obtained drainage densities d_s (i.e. areas of drained roads and surface waters per crop area) for each study site. The average drainage densities for arable land and vineyard sites are provided in Table 3 for the reference parameter set and the two extreme parameter sets. For all types of non-target areas, the drainage densities in vineyards are by a factor two to three higher than in arable land. This indicates that the spray drift potential in vineyard sites is higher than in arable land sites, independent of the spraying method used. Drained roads are responsible for around 73% to 84% of the total drainage density for both crop types. These results are similar to a modelling study of Alder et al. (2015), reporting that 71% of the total drainage density is caused by drained roads.

Table 3
Average drainage densities d_s of arable land and vineyard sites obtained from the reference parameter set p_{ref}. In brackets, the results for the extreme parameter sets (p_{min}, p_{max}) are given.

Crop type	Roads drained to shortcuts d_{RSC}	Roads drained to surface waters d_{RSW}	All drained roads d_R	Surface waters d_{SW}	Total drainage density
Arable land	1.4% [0.81%; 2.6%]	0.11% [0.07%; 0.17%]	1.5% [0.88%; 2.8%]	0.28% [0.28%; 0.51%]	1.8% [1.2%; 3.3%]
Vineyards	4.2% [1.6%; 6.0%]	0.23% [0.28%; 0.98%]	4.4% [1.9%; 7.0%]	0.63% [0.63%; 2.6%]	5.1% [2.5%; 9.6%]

3.2. Spray drift losses to drained roads and surface waters

3.2.1. Model output example

From the spray drift model, we obtained estimates for the fraction of the applied amount lost via drift to each non-target area raster cell (either drained roads, or surface waters). In Fig. 4, the spray drift model output is depicted on the example of the study site Clarmont, the wind direction southwest (SW), and the reference parameter set. The depicted part of the study site illustrates classical spray drift patterns that were also found frequently in the other study sites. Many roads are drained by storm drainage inlets (A, B) and the drift deposited per area is much higher for these roads than for surface waters (e.g. A vs. D). This can be explained by two reasons: First, drained roads are mostly situated much closer to crop areas, and they are not protected by riparian forests. Second, as mentioned in the previous section, the drainage densities are much higher for drained roads than for surface waters (Section 3.1). These factors lead to a much higher total spray drift deposition on drained roads compared to surface waters. However, this does not mean that all roads have a high potential for spray drift wash-off to surface waters. The depicted part of the study site also shows examples of drained roads receiving significantly less drift. This either is caused by larger distances between the road and the next sprayed plot along the wind line (B), by barriers that intercept spray drift (forest, hedges, trees, buildings; no example shown for wind direction southwest) or since the road is classified as undrained (C). Although undrained roads also receive spray drift from the adjacent plots, the washed off runoff is expected to infiltrate in the adjacent agricultural areas. The model results also show that depending on the wind direction, the spray drift deposition on non-target areas can vary strongly at the local scale. For example, the road areas marked with the letter B, would receive much more spray drift for the wind direction east compared to the depicted wind direction southwest.

3.2.2. Losses for all study sites

The modelled spray drift losses to different non-target areas are shown in Fig. 5 for arable land sites, and in Fig. 6 for vineyards. In vineyards, the total drift losses f_{lost} to drained roads and surface waters range between 0.063% and 0.20% on average, depending on the model parametrisation (Table 4). Almost all of these losses are deposited on drained roads. These results align well with measurements in a French vineyard catchment (Lefrancq et al., 2013) where spray drift deposition on roads amounted to between 0.07% and 0.57% of the applied amount.

Compared to vineyard sites, the average spray drift deposition to drained roads and surface waters in arable land sites is much lower, equalling between 0.0015% and 0.0049% of the applied amount. With the exception of the site Bex, all vineyard sites show larger total spray

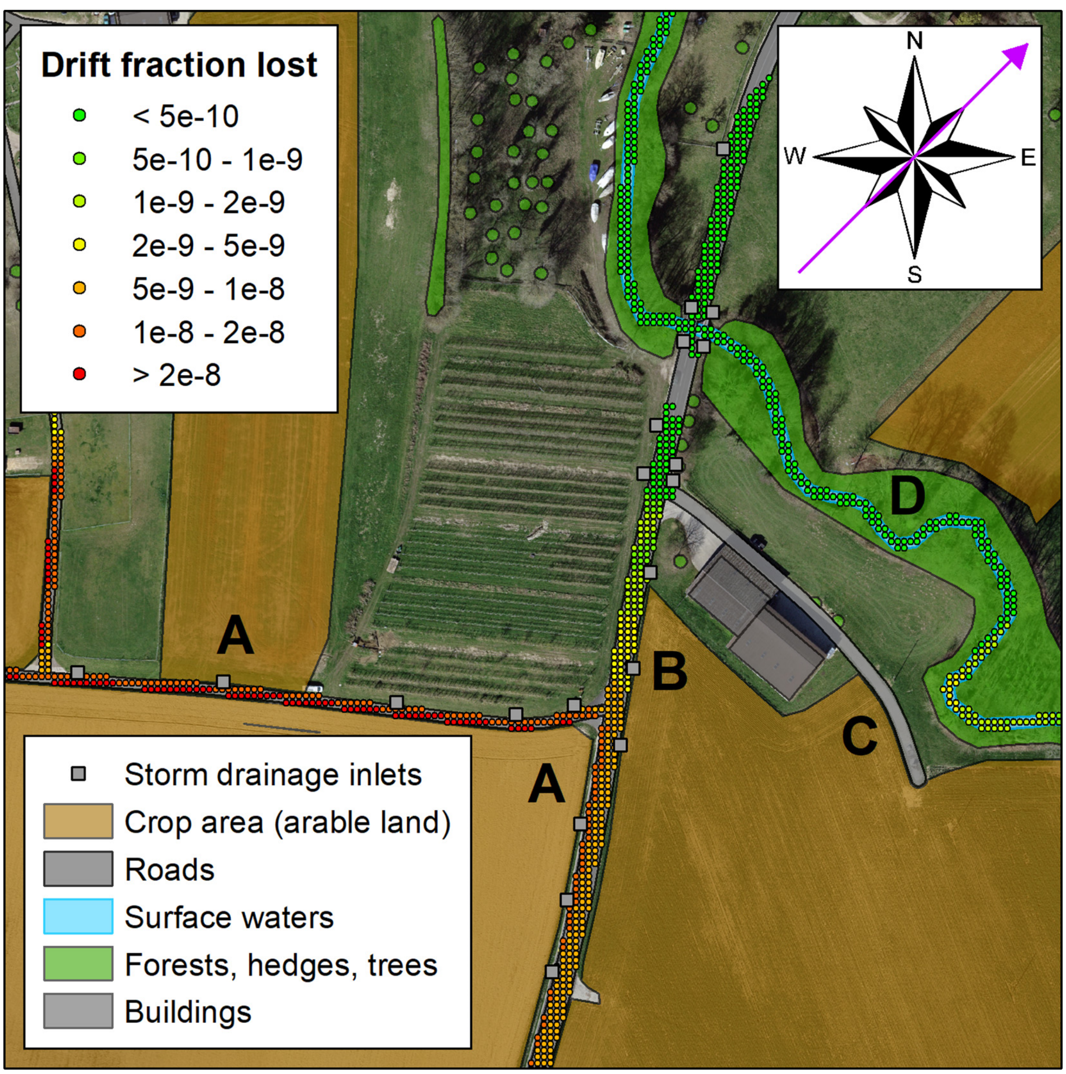

Fig. 4. Spray drift model results for the reference parameter set and the wind direction southwest, for a part of the study site Clarmont. The values reported represent the fraction of drift lost to each non-target area cell relative to the total amount applied in the whole study site. In the center of the image, an orchard is situated. Since in this study only arable land and vineyards were analysed, orchard areas were not used for modelling and considered as an empty area. Sources: Kanton Aargau et al. (2020); Swisstopo (2019); Swisstopo (2020b).

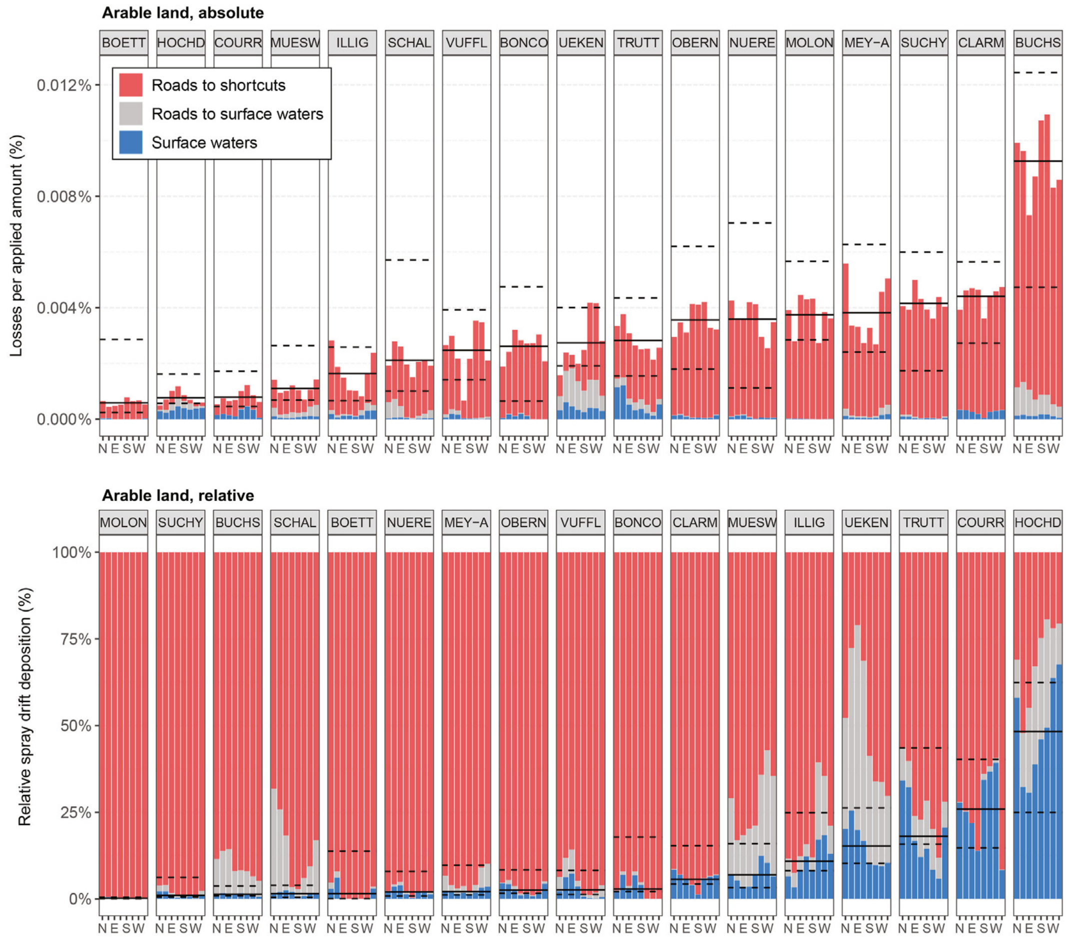

Fig. 5. Drift losses in arable land study sites per wind direction: (A) Fraction lost per total amount applied on arable crops ($f_{lost,w}$). Red bars represent the drift to roads draining via shortcuts to surface waters, grey bars represent the drift to roads directly draining to surface waters, and blue bars represent direct drift to surface waters. The black solid lines indicate the average losses over all wind directions resulting from the reference parameter set (p_{ref}). The dashed lines report the average losses from the extreme parameter sets (p_{min}, p_{max}). (B) Losses per non-target area, relative to the losses to all three non-target areas ($f_{lost,rel,w}$). The black solid lines indicate the average losses to surface waters over all wind directions, resulting from the reference parameter set (p_{ref}). The dashed lines represent the average losses to surface waters resulting from the extreme parameter sets ($p_{SWrel,min}$, $p_{SWrel,max}$).

drift losses than each of the arable land sites. This difference can be explained by higher drainage densities in vineyards (see Section 3.1) and by the different application method used in vineyards (air blast sprayers instead of boom sprayers). It remains unclear, to which degree the spatial relationship between non-target areas, roads, and barriers additionally influences this result. In the study site Bex, the majority of the storm

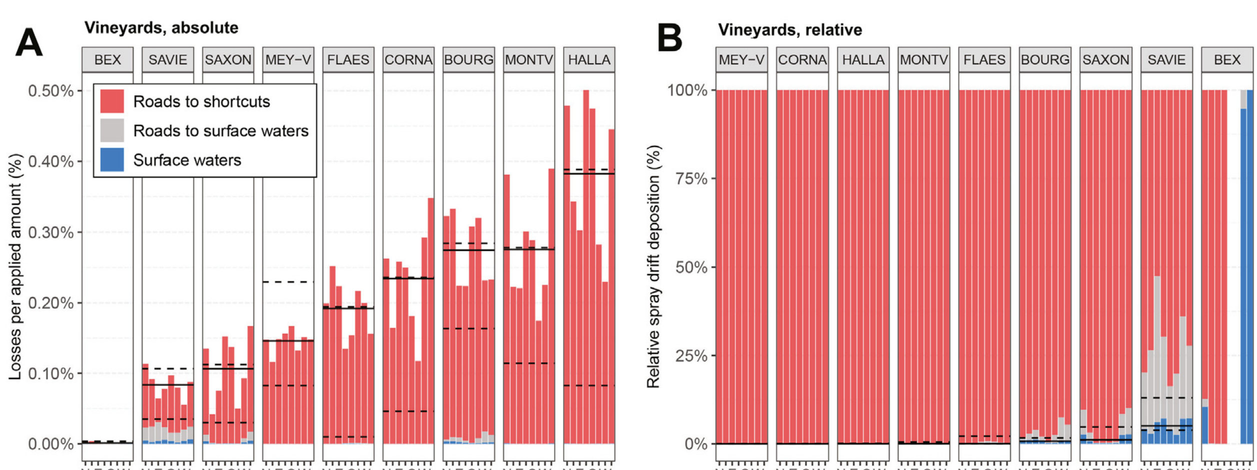

Fig. 6. Drift losses in vineyard study sites per wind direction: (A) Fraction lost per total amount applied in vineyards ($f_{lost,w}$). Red bars represent the drift to roads draining via shortcuts to surface waters, grey bars represent the drift to roads directly draining to surface waters, and blue bars represent direct drift to surface waters. The black solid lines indicate the average losses over all wind directions resulting from the reference parameter set (p_{ref}). The dashed lines report the average losses from the extreme parameter sets (p_{min}, p_{max}). (B) Losses per non-target area, relative to the losses to all three non-target areas ($f_{lost,rel,w}$). The black solid lines indicate the average losses to surface waters over all wind directions, resulting from the reference parameter set (p_{ref}). The dashed lines represent the average losses to surface waters resulting from the extreme parameter sets ($p_{SWrel,min}$, $p_{SWrel,max}$).

Table 4
Average drift to drained roads and surface waters for arable land and vineyard sites. The reported values indicate the results of the reference parameter set. In brackets, the results of the extreme parameter sets are given. For the calculation of relative losses in vineyard sites, the study site Bex was excluded.

Target area	All drained roads		Surface waters	
	Losses per applied amount $f_{lost,R}$ (−)	Relative loss on non-target area type $f_{lost,rel,RSC} + f_{lost,rel,RSW}$	Losses per applied amount $f_{lost,SW}$ (−)	Relative loss on non-target area type $f_{lost,rel,SW}$
Arable land	$2.8 \cdot 10^{-5}$ [$1.4 \cdot 10^{-5}$; $4.6 \cdot 10^{-5}$]	91.3% [81.8%; 94.7%]	$1.5 \cdot 10^{-6}$ [$1.3 \cdot 10^{-6}$; $2.9 \cdot 10^{-6}$]	8.7% [5.3%; 18.2%]
Vineyards	$1.8 \cdot 10^{-3}$ [$6.2 \cdot 10^{-4}$; $2.0 \cdot 10^{-3}$]	99.1% [97.2%; 99.3%]	$9.0 \cdot 10^{-6}$ [$8.7 \cdot 10^{-6}$; $1.2 \cdot 10^{-5}$]	0.9% [0.7%; 2.8%]

drainage system of vineyard roads drains to an infiltration basin. The density of roads drained to surface waters is therefore much smaller than in other study sites. Moreover, the closest surface waters are located far away from the vineyards. These two factors lead to very low drift losses to drained roads and surface waters in Bex.

As mentioned in Section 3.2.1, spray drift losses can vary strongly on the local scale depending on wind direction. However, our model results show that this variation is also observed at the catchment scale. These differences can amount up to a factor 4 in vineyard sites (Fig. 6), and a factor 5 in arable land sites (Fig. 5). On average, the difference between the wind direction with the highest and lowest spray drift deposition equals a factor of 2.2. For certain study sites, spray drift deposition on drained roads and surface waters could therefore be reduced considerably by applying pesticides during favourable wind directions.

A comparison of the relative spray drift losses to drained roads and to surface waters reveals that most of the spray drift is deposited on drained roads (average – 82 to 95% for arable land, 97 to 99% for vineyards; Table 4). Accordingly, the spray drift deposited on drained roads that potentially can be washed off to surface waters is much larger than the spray drift directly deposited in surface waters for both crop types, provided that the legally required buffer widths are kept (see discussion in next paragraph). As shown in Figs. 5 and 6, most of the spray drift deposition on drained roads is taking place on roads drained by shortcuts. Shortcuts therefore strongly increase the potential of spray drift wash-off from roads to surface waters.

In our analysis, we assumed that farmers comply with the buffer widths according to Swiss regulations and to Swiss proof of ecological performance (see Section 2.2.3). However, the buffer widths to surface waters are often not kept in Swiss vineyards. Therefore, the above-mentioned results represent an ideal situation for the drift to surface waters and the real drift to surface waters is higher.

In the timespan between the application and the next rain event, different degradation processes but also sorption may lead to a significant reduction of the spray drift available for wash-off from roads (Jiang and Gan, 2016). Little is known on the degradation of pesticides on concrete and asphalt surfaces. The available literature suggests that pesticide degradation on concrete surfaces is mainly driven by photolysis, oxidation, and hydrolysis, enhanced by the strong alkalinity of concrete (Richards et al., 2017; Thuyet et al., 2012). Additionally, it was shown that the degradation of pesticides on concrete surfaces may lead to the formation of biologically active transformation products (Jiang and Gan, 2016; Richards et al., 2017). Road runoff containing only pesticide transformation products may therefore still lead to adverse effects to aquatic species when it is discharged to surface waters.

During experiments on concrete and asphalt roads, 57% or more of the amount applied of substances with low soil adsorption coefficients ($K_{OC} < 250$ mL g^{-1}) was washed off (Ramwell et al., 2002; Thuyet et al., 2012). In these studies, the time between application and rainfall amounted between six hours and seven days, being a realistic range for the time elapsing between application and rainfall in many parts of Western and Central Europe. The road data used within this study allowed us to separate the spray drift deposition between different types of road surfaces. This analysis revealed that the majority of spray drift to drained roads (75%) in the study sites is deposited on asphalt or concrete roads. For substances with low K_{OC}, we therefore expect the amount of spray drift washed off from drained roads to clearly exceed the amount of spray drift directly deposited in streams. In contrast, for substances with higher K_{oc}, maximal wash-off reported during the first rainfall events after application reached up to 5.8% (Thuyet et al., 2012) and 2.7% (Jiang et al., 2012). For these substances, we therefore expect the amount of spray drift washed off from roads to surface waters to be in the same order of magnitude or lower than the direct spray drift to surface waters. However, it should be kept in mind that such substances might still be washed off during later rain events with the road acting as a pesticide reservoir (Jiang et al., 2010; Jiang et al., 2012).

To determine the relevance of spray drift wash-off from drained roads for the total pesticide load in the stream, we compared the spray drift losses to total loss rates to surface waters. Total losses to surface waters typically range between 0.005% and 1% of the applied amount (Doppler et al., 2014; Leu et al., 2004; Riise et al., 2004; Siimes et al., 2006). Therefore, in arable land sites, the spray drift losses to drained roads (0.0015% and 0.0049%) are small compared to typical total loss rates. However, in vineyard sites, the losses to drained roads (0.063% to 0.20%) represent a major fraction compared to typical total loss rates. For substances with low K_{OC} applied in vineyards, we therefore expect the wash-off from drained roads to be a relevant transport pathway compared to total pesticide losses to surface waters.

In the above-mentioned studies (Doppler et al., 2014; Leu et al., 2004; Riise et al., 2004; Siimes et al., 2006), total loss rates were (with one exception) only determined for herbicides. Because of their physico-chemical properties and the timing of application, herbicides may not well represent the situation for fungicides and insecticides. For example, the average K_{OC} of herbicides authorized in Switzerland (92 mL g^{-1}), is much lower than for fungicides (1100 mL g^{-1}) and insecticides (1700 mL g^{-1}) (see Section A2.1). These two groups are applied at later growth stages when the crop interception is higher and less surface runoff is formed on the field. Due to their high K_{OC} and the timing of application their losses by surface runoff formed on the field and by tile drainage transport are expected to be reduced relative to herbicides. Therefore, one may expect that spray drift wash-off of fungicides and insecticides contributes more to their total losses compared to herbicides. Consequently, even for substances with high K_{OC} (such as most fungicides and insecticides), losses via spray drift wash-off from roads might be a relevant pathway in relation to their total loss rates.

Furthermore, it is important to note that between rain events spray drift losses to drained roads are accumulating and are then all washed off at once. This might lead to much higher concentration peaks than direct spray drift deposition to surface waters during single spray applications.

Previous studies have shown that spray drift to roads and subsequent wash-off can be an important transport pathway in single catchments (Ammann et al., 2020; Lefrancq et al., 2014; Rübel, 1999). Our results indicate that for catchments with high densities of drained roads and for application methods with a high spray drift potential, these findings can be generalized.

3.3. Model uncertainties

In the previous section, model uncertainty was addressed by reporting the results as a range between the minimal and maximal parameter sets (p_{min}, p_{max} ; $p_{SWrel,min}$, $p_{SWrel,max}$). In the following, we elaborate on the importance of single model parameters on the overall uncertainty and on additional uncertainties related to the models used in this study.

The combined sensitivity analysis of the surface runoff connectivity and the spray drift model shows that the parameters road carving depth d_{road} and infiltration distance d_{inf} cause the largest model uncertainties (details – Section A2.2). These two parameters are both used to classify roads as drained or undrained in the surface runoff connectivity model. This indicates that the classification of roads is one of the major uncertainty factors. To check the plausibility of road classification, we therefore compared the areas classified as drained roads to flow paths mapped during a snowmelt event on 12 March 2018 in the study site Schalunen (Schönenberger et al., in review). This comparison suggests that the road areas drained by shortcuts are underestimated by the reference parameter set and that they are rather in the range of the values resulting from the maximal total drift parameter set (p_{max}). However, during this snowmelt event, the amount of runoff on roads was exceptionally high. Accordingly, we expect that flow paths were longer during the snowmelt event than during most rain events. Therefore, this comparison affirms the plausibility of the range of model outputs. Nevertheless, the classification of roads as drained or undrained remains a major source of uncertainty. Further studies on spray drift wash-off from roads should therefore aim on validating the surface runoff connectivity model and the classification of roads, for example by extensive mapping of flow paths during rain events. A detailed discussion on how surface runoff connectivity models could be validated is provided in the publication describing the original version of the connectivity model used here (Schönenberger and Stamm, 2021).

Additional uncertainties are caused by the extrapolation of the spray drift curves to a minimal drift distance of 0.5 m. During the field trials used for spray drift curve determination, the minimal drift distances measured were 1 m (arable land) and 3 m (vineyards) from the sprayed plot. Since the buffer width around surface waters equals 6 m, this extrapolation was only used for estimating spray drift to drained roads, but not to surface waters. If the effective drift curve is below the extrapolated drift curve (Eq. (2)) for distances shorter than the minimal measured distances, our model would lead to an overestimation of the spray drift to drained roads. To ensure that our conclusions are not an artefact of the spray drift curve extrapolation, we performed another model run using the reference parameter set. However, for distances smaller than the minimal measured distances, we did not use the extrapolated spray drift curve, but restricted the spray drift deposition to the values at the minimal measured distance (1 m/3 m). For arable land sites, this led to a reduction of only 2.5% of the estimated drift losses to drained roads. The extrapolation uncertainty can therefore be neglected for this crop type. However, for vineyards, a much larger extrapolation uncertainty (reduction of 51%) was found. This uncertainty is not large enough to change the conclusions drawn on the potential of spray drift wash-off from drained roads in vineyards (Section 3.2.2). However, to reduce the uncertainty in the estimation of spray drift deposition on vineyard roads, additional drift trials in ultimate proximity of vineyard plots (< 3 m) would be needed. Furthermore, it should be mentioned that during the spray drift trials in vineyards (Rautmann et al., 1999) the wind direction was always parallel to the blowing direction of the air blast sprayers. However, this is not the case in reality. Therefore, in reality, the effective spray drift deposition might be lower than reported for non-target areas lying in the direction of prevalent wind, but higher for non-target areas lying in other directions.

In this study, we assumed that pesticides are not applied within a buffer of 6 m around surface waters. However, several pesticides are only authorized for usage outside of larger buffers (20 m, 50 m, or 100 m). For these pesticides, the direct drift to surface waters is much lower and the relative importance of drift wash-off from roads is much higher. This further underlines the high potential of spray drift wash-off from roads compared to direct drift to surface waters.

3.4. Implications for practice

The results presented in this study suggest that spray drift wash-off from drained roads is a major source for the pesticide pollution of surface waters, at least in vineyards and for pesticides with low K_{OC}. To reduce spray drift to drained roads, various measures could be worth considering. These measures include drift reducing spraying techniques, and drift barriers or buffer strips between the sprayed plots and drained roads. Additionally, precision farming technologies could further help reducing spray drift to drained roads. For example, GPS-assisted sprayers could be programmed to automatically turn off spray nozzles when they get too close to a drained road.

We used the spray drift model presented here to assess the potential of buffer strips for reducing spray drift to drained roads. For the reference parameter set, our model predicts that a 3 m buffer around all drained roads would lead to a 37% and 74% reduction of spray drift in arable land, and vineyard sites, respectively. With a 6 m buffer, spray drift to drained roads would be reduced by 56% and 90%. However, it has to be kept in mind that especially for vineyards, the spray drift curves are rather uncertain for distances smaller than 3 m (see Section 3.3).

Spray drift to drained roads and subsequent wash-off is currently not considered in European or Swiss pesticide authorisation and legislation. Our results however indicate that this transport pathway is relevant, at least in certain cases. This demonstrates that current regulations only cover a part of the total pesticide transport to surface waters related to spray drift. The same issue has been shown for the surface runoff related transport of pesticides via shortcuts (Schönenberger et al., in review; Schönenberger and Stamm, 2021). Authorities should therefore consider the potential of pesticide transport via shortcuts in the pesticide registration process and when designing regulations. At the same time, farmers should be aware of the potential of this process when applying pesticides.

4. Conclusions

- In agricultural catchments in Switzerland, many roads are drained by shortcuts (storm drainage system inlets, channel drains, ditches) or directly to surface waters. The density of such roads is 2.7 to 7 times larger than the density of surface waters.
- The amount of spray drift deposited on drained roads is much larger than the direct drift deposition in surface waters. In the arable land sites studied, spray drift to drained roads exceeded the direct drift by a factor of 4.5 to 18. In vineyard sites, this factor amounts between 35 and 140, assuming that farmers comply with the legally required buffer widths. Most spray drift losses to drained roads are deposited on roads drained by shortcuts, and only a minor part is deposited on roads directly drained to surface waters.
- Compared to typical total pesticide loss rates to surface waters, the spray drift losses to drained roads are rather small in arable land sites (losses equal between 0.0015% and 0.0049% of the applied amount). However, the losses to drained roads in vineyard sites losses to drained roads are substantial (0.063% to 0.20% of the applied amount).
- Current literature suggests that major fractions of the spray drift on roads can be washed off during subsequent rain events, especially for (but not restricted to) substances with low soil adsorption coefficients (K_{OC}). Especially in vineyards, the spray drift wash-off from drained roads is therefore expected to be a relevant transport pathway for pesticides to surface waters.
- These findings should be considered for adapting pesticide registration procedures and for implementing best management practices in critical agricultural areas such as vineyards.

CRediT authorship contribution statement

Urs Schönenberger: Conceptualization, Methodology, Software, Investigation, Formal analysis, Data curation, Writing – original draft, Visualization. **Janine Simon:** Investigation, Data curation, Writing – review & editing. **Christian Stamm:** Conceptualization, Writing – review & editing, Funding acquisition, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary information

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References

Aargau, Kanton, Genf, Kanton, Jura, Kanton, Luzern, Kanton, Schaffhausen, Kanton, Thurgau, Kanton, 2020. Landwirtschaftliche Bewirtschaftung: Nutzungsflächen. Konferenz der kantonalen Geoinformations- und Katasterstellen. https://www.geodienste.ch/services/lwb_nutzungsflaechen. (Accessed 28 October 2021).

Alder, S., Prasuhn, V., Liniger, H., Herweg, K., Hurni, H., Candinas, A., et al., 2015. A high-resolution map of direct and indirect connectivity of erosion risk areas to surface waters in Switzerland-a risk assessment tool for planning and policy-making. Land Use Policy 48, 236–249. https://doi.org/10.1016/j.landusepol.2015.06.001.

Ammann, L., Doppler, T., Stamm, C., Reichert, P., Fenicia, F., 2020. Characterizing fast herbicide transport in a small agricultural catchment with conceptual models. J. Hydrol. 586. https://doi.org/10.1016/j.jhydrol.2020.124812.

Bonzini, S., Verro, R., Otto, S., Lazzaro, L., Finizio, A., Zanin, G., et al., 2006. Experimental validation of a geographical information systems-based procedure for predicting pesticide exposure in surface water. Environ. Sci. Technol. 40, 7561–7569. https://doi.org/10.1021/es0615324.

Carlsen, S.C.K., Spliid, N.H., Svensmark, B., 2006. Drift of 10 herbicides after tractor spray application. 2. Primary drift (droplet drift). Chemosphere 64, 778–786. https://doi.org/10.1016/j.chemosphere.2005.10.060.

ChemRRV, 2005. Verordnung zur Reduktion von Risiken beim Umgang mit bestimmten besonders gefährlichen Stoffen, Zubereitungen und Gegenständen. SR 814.81. Der Schweizerische Bundesrat, pp. 1–178.

Dehotin, J., Breil, P., Braud, I., de Lavenne, A., Lagouy, M., Sarrazin, B., 2015. Detecting surface runoff location in a small catchment using distributed and simple observation method. J. Hydrol. 525, 113–129. https://doi.org/10.1016/j.jhydrol.2015.02.051.

Doppler, T., Camenzuli, L., Hirzel, G., Krauss, M., Lück, A., Stamm, C., 2012. Spatial variability of herbicide mobilisation and transport at catchment scale: insights from a field experiment. Hydrol. Earth Syst. Sci. 16, 1947–1967. https://doi.org/10.5194/hess-16-1947-2012.

Doppler, T., Luck, A., Camenzuli, L., Krauss, M., Stamm, C., 2014. Critical source areas for herbicides can change location depending on rain events. Agric. Ecosyst. Environ. 192, 85–94. https://doi.org/10.1016/j.agee.2014.04.003.

DZV, 2013. Verordnung über die Direktzahlungen an die Landwirtschaft. SR 910.13. Der Schweizerische Bundesrat, pp. 1–150.

Fiener, P., Auerswald, K., Van Oost, K., 2011. Spatio-temporal patterns in land use and management affecting surface runoff response of agricultural catchments-a review. Earth Sci. Rev. 106, 92–104. https://doi.org/10.1016/j.earscirev.2011.01.004.

FOEN, 2012. Topographical Catchment Areas of Swiss Waterbodies 2 km^2. Federal Office for the Environment, Bern. https://opendata.swiss/en/perma/6d9c8ba5-2532-46ed-bc26-0a4017787a56@bundesamt-fur-umwelt-bafu. (Accessed 28 October 2021).

FSO, 2014. Swiss Land Use Statistics Nomenclature 2004 - Metainformmation on Geodata. 48. Federal Statistical Office, Neuchâtel, Switzerland.

Ganzelmeier, H., 1995. Studies on the Spray Drift of Plant Protection Products Results of a Test Program Carried Out Throughout the Federal Republic of Germany. Blackwell Wissenschafts-Verlag, Berlin.

Gassmann, M., Stamm, C., Olsson, O., Kümmerer, K., Weiler, M., 2013. Model-based estimation of pesticides and transformation products and their export pathways in a headwater catchment. Hydrol. Earth Syst. Sci. 17, 5213–5228. https://doi.org/10.5194/hess-17-5213-2013.

Golla, B., Strassemeyer, J., Koch, H., Rautmann, D., 2011. Eine Methode zur stochastischen Simulation von Abdriftwerten als Grundlage für eine georeferenzierte probabilistische Expositionsabschätzung von Pflanzenschutzmitteln. J. Kult. 63, 33–44.

Heathwaite, A.L., Quinn, P.F., Hewett, C.J.M., 2005. Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. J. Hydrol. 304, 446–461. https://doi.org/10.1016/j.jhydrol.2004.07.043.

Holterman, H.J., Van de Zande, J.C., 2008. The Cascade Drift Module: a GIS-based study on regional pesticide deposition. Conference: International Advances in Pesticide Application. 84. Association of Applied Biologists, Warwick, UK, pp. 83–90 Cambridge, UK.

Huber, A., Bach, M., Frede, H.G., 2000. Pollution of surface waters with pesticides in Germany: modeling non-point source inputs. Agric. Ecosyst. Environ. 80, 191–204. https://doi.org/10.1016/S0167-8809(00)00145-6.

Jensen, P.K., Olesen, M.H., 2014. Spray mass balance in pesticide application: a review. Crop Prot. 61, 23–31. https://doi.org/10.1016/j.cropro.2014.03.006.

Jiang, W., Gan, J., 2016. Conversion of pesticides to biologically active products on urban hard surfaces. Sci. Total Environ. 556, 63–69. https://doi.org/10.1016/j.scitotenv.2016.02.165.

Jiang, W., Lin, K., Haver, D., Qin, S., Ayre, G., Spurlock, F., et al., 2010. Wash-off potential of urban use insecticides on concrete surfaces. Environ. Toxicol. Chem. 29, 1203–1208. https://doi.org/10.1002/etc.184.

Jiang, W.Y., Haver, D., Rust, M., Gan, J., 2012. Runoff of pyrethroid insecticides from concrete surfaces following simulated and natural rainfalls. Water Res. 46, 645–652. https://doi.org/10.1016/j.watres.2011.11.023.

de Jong, F.M.W., de Snoo, G.R., van de Zande, J.C., 2008. Estimated nationwide effects of pesticide spray drift on terrestrial habitats in the Netherlands. J. Environ. Manag. 86, 721–730. https://doi.org/10.1016/j.jenvman.2006.12.031.

Julius Kühn-Institut, 2013. Richtlinie für die Prüfung von Pflanzenschutzgeräten - Messung der direkten Abdrift beim Ausbringen von flüssigen Pflanzenschutzmitteln im Freiland, pp. 1–7 7-1.5. Julius Kühn-Institut, Braunschweig.

Kladivko, E.J., Brown, L.C., Baker, J.L., 2001. Pesticide transport to subsurface tile drains in humid regions of North America. Crit. Rev. Environ. Sci. Technol. 31, 1–62. https://doi.org/10.1080/20016491089163.

Kubiak, R., Hommen, U., Bach, M., Classen, S., Gergs, A., Golla, B., 2014. Georeferenced Probabilistic Risk Assessment of Pesticides - Further Advances in Assessing the Risk to Aquatic Ecosystems by Spray Drift from Permanent Crops. 05/2014. Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety, Dessau-Roßlau. http://www.umweltbundesamt.de/publikationen/georeferenced-probabilistic-risk-assessment-of. (Accessed 28 October 2021).

Larsbo, M., Sandin, M., Jarvis, N., Etana, A., Kreuger, J., 2016. Surface runoff of pesticides from a clay loam field in Sweden. J. Environ. Qual. 45, 1367–1374. https://doi.org/10.2134/jeq2015.10.0528.

Lefrancq, M., Imfeld, G., Payraudeau, S., Millet, M., 2013. Kresoxim methyl deposition, drift and runoff in a vineyard catchment. Sci. Total Environ. 442, 503–508. https://doi.org/10.1016/j.scitotenv.2012.09.082.

Lefrancq, M., Payraudeau, S., García Verdú, A.J., Maillard, E., Millet, M., Imfeld, G., 2014. Fungicides transport in runoff from vineyard plot and catchment: contribution of non-target areas. Environ. Sci. Pollut. Res. 21, 4871–4882. https://doi.org/10.1007/s11356-013-1866-8.

Leu, C., Singer, H., Stamm, C., Müller, S.R., Schwarzenbach, R.P., 2004. Simultaneous assessment of sources, processes, and factors influencing herbicide losses to surface waters in a small agricultural catchment. Environ. Sci. Technol. 38, 3827–3834. https://doi.org/10.1021/es0499602.

Linders, J., Adriaanse, P., Allen, R., Capri, E., Gouy, V., Hollis, J., 2003. FOCUS Surface Water Scenarios in the EU Evaluation Process Under 91/414/EEC. 2. Report prepared by the FOCUS Working group on surface water Scenarios. European Commission SANCO/4802/2001–rev.

Padovani, L., Trevisan, M., Capri, E., 2004. A calculation procedure to assess potential environmental risk of pesticides at the farm level. Ecol. Indic. 4, 111–123. https://doi.org/10.1016/j.ecolind.2004.01.002.

Payraudeau, S., Junker, P., Imfeld, G., Gregoire, C., 2009. Characterizing hydrological connectivity to identify critical source areas for pesticides losses. 18^{th} World Imacs Congress and Modsim09, 13-17 July 2009. International Congress on Modelling and Simulation, Cairns, Australia, pp. 1879–1885.

Ramwell, C.T., 2005. Herbicide sorption to concrete and asphalt. Pest Manag. Sci. 61, 144–150. https://doi.org/10.1002/ps.959.

Ramwell, C.T., Heather, A.I.J., Shepherd, A.J., 2002. Herbicide loss following application to a roadside. Pest Manag. Sci. 58, 695–701. https://doi.org/10.1002/ps.506.

Rautmann, D., Streloke, M., Winkler, R., 1999. New basic drift values in the authorization procedure for plant protection products. In: Forster, R., Streloke, M. (Eds.), Workshop on Risk Assessment and Risk Mitigation Measures in the Context of the Authorization of Plant Protection Products (WORMM). 383, Braunschweig, Germany, pp. 133–141.

Reichenberger, S., Bach, M., Skitschak, A., Frede, H.G., 2007. Mitigation strategies to reduce pesticide inputs into ground- and surface water and their effectiveness; a review. Sci. Total Environ. 384, 1–35. https://doi.org/10.1016/j.scitotenv.2007.04.046.

Richards, J., Lu, Z., Fu, Q., Schlenk, D., Gan, J., 2017. Conversion of pyrethroid insecticides to 3-phenoxybenzoic acid on urban hard surfaces. Environ. Sci. Technol. Lett. 4 (12), 546–550. https://doi.org/10.1021/acs.estlett.7b00466.

Riise, G., Lundekvam, H., Wu, Q.L., Haugen, L.E., Mulder, J., 2004. Loss of pesticides from agricultural fields in SE Norway – runoff through surface and drainage water. Environ. Geochem. Health 26, 269–276. https://doi.org/10.1023/B:EGAH.0000039590.84335.d6.

Röpke, B., Bach, M., Frede, H.G., 2004. DRIPS - a DSS for estimating the input quantity of pesticides for German river basins. Environ. Model Softw. 19, 1021–1028. https://doi.org/10.1016/j.envsoft.2003.11.005.

Rübel, A., 1999. Eintrag von Pflanzenschutzmitteln in Oberflächengewässer durch den Weinbau in Steillagen. Abteilung Hydrologie. PhD. Universität Trier, p. 176.

Sandin, M., Piikki, K., Jarvis, N., Larsbo, M., Bishop, K., Kreuger, J., 2018. Sci. Total Environ. 610, 623–634. https://doi.org/10.1016/j.scitotenv.2017.08.068.

Schad, T., Schulz, R., 2011. Xplicit, a novel approach in probabilistic spatiotemporally explicit exposure and risk assessment for plant protection products. Integr. Environ. Assess. Manag. 7, 612–623. https://doi.org/10.1002/ieam.205.

Schönenberger, U., Dax, A., Beck, B., Vogler, B., Stamm, C., 2021. Pesticide concentrations in hydraulic shortcuts of a small Swiss agricultural catchment (in review).

Schönenberger, U., Stamm, C., 2021. Hydraulic shortcuts increase the connectivity of arable land areas to surface waters. Hydrol. Earth Syst. Sci. 25, 1727–1746. https://doi.org/10.5194/hess-25-1727-2021.

Schulz, R., 2001. Comparison of spray drift- and runoff-related input of azinphos-methyl and endosulfan from fruit orchards into the Lourens River, South Africa. Chemosphere 45 (4–5), 543–551. https://doi.org/10.1016/S0045-6535(00)00601-9.

Schwarzenbach, R.P., Escher, B.I., Fenner, K., Hofstetter, T.B., Johnson, C.A., von Gunten, U., et al., 2006. The challenge of micropollutants in aquatic systems. Science 313, 1072–1077. https://doi.org/10.1126/science.1127291.

Siimes, K., Rämö, S., Welling, L., Nikunen, U., Laitinen, P., 2006. Comparison of the behaviour of three herbicides in a field experiment under bare soil conditions. Agric. Water Manag. 84, 53–64. https://doi.org/10.1016/j.agwat.2006.01.007.

Simon, J., 2019. Pestizideintrag in Oberflächengewässer via Kurzschlüsse in Rebbergen. Professur für Hydrologie. Bachelor of Science. Albert-Ludwigs-Universität Freiburg i. Br., Freiburg im Breisgau, p. 27.

Spycher, S., Mangold, S., Doppler, T., Junghans, M., Wittmer, I., Stamm, C., et al., 2018. Pesticide risks in small streams-how to get as close as possible to the stress imposed on aquatic organisms. Environ. Sci. Technol. 52, 4526–4535. https://doi.org/10.1021/acs.est.8b00077.

Spycher, S., Teichler, R., Vonwyl, E., Longrée, P., Stamm, C., Singer, H., 2019. Anhaltend hohe PSM-Belastung in Bächen. NAWA SPEZ 2017: kleine Gewässer in Gebieten mit intensiver Landwirtschaft verbreitet betroffen. Aqua & Gas 99, 14–25.

Stehle, S., Schulz, R., 2015. Agricultural insecticides threaten surface waters at the global scale. Proc. Natl. Acad. Sci. U. S. A. 112, 5750–5755. https://doi.org/10.1073/pnas.1500232112.

Stephenson, G.R., Ferris, I.G., Holland, P.T., Nordberg, M., 2006. Glossary of terms relating to pesticides (IUPAC Recommendations 2006). Pure Appl. Chem. 78, 2075–2154. https://doi.org/10.1351/pac200678112075.

Swisstopo, 2010. In: Swisstopo (Ed.), swissTLM3D – The Topographic Landscape Model. Federal Office of Topography, Wabern, Switzerland.

Swisstopo, 2019. In: Swisstopo (Ed.), SWISSIMAGE – The Digital Color Orthophotomosaic of Switzerland. Federal Office of Topography, Wabern.

Swisstopo, 2020a. In: Swisstopo (Ed.), swissALTI3D – The High Precision Digital Elevation Model of Switzerland. Federal Office of Topography, Wabern, Switzerland.

Swisstopo, 2020b. In: Swisstopo (Ed.), swissTLM3D – The Topographic Landscape Model. Federal Office of Topography, Wabern, Switzerland.

Tarboton, D.G., 1997. A new method for the determination of flow directions and upslope areas in grid digital elevation models. Water Resour. Res. 33, 309–319. https://doi.org/10.1029/96wr03137.

Thuyet, D.Q., Jorgenson, B.C., Wissel-Tyson, C., Watanabe, H., Young, T.M., 2012. Wash off of imidacloprid and fipronil from turf and concrete surfaces using simulated rainfall. Sci. Total Environ. 414, 515–524. https://doi.org/10.1016/j.scitotenv.2011.10.051.

Travis, K.Z., Hendley, P., 2001. Probabilistic risk assessment of cotton pyrethroids: IV. Landscape-level exposure characterization. Environ. Toxicol. Chem. 20, 679–686. https://doi.org/10.1002/etc.5620200329.

Ucar, T., Hall, F.R., 2001. Windbreaks as a pesticide drift mitigation strategy: a review. Pest Manag. Sci. 57, 663–675. https://doi.org/10.1002/ps.341.

Viret, O., Siegfried, W., Holliger, E., Raisigl, U., 2003. Comparison of spray deposits and efficacy against powdery mildew of aerial and ground-based spraying equipment in viticulture. Crop Prot. 22, 1023–1032. https://doi.org/10.1016/S0261-2194(03)00119-4.

Wang, M., Rautmann, D., 2008. A simple probabilistic estimation of spray drift–factors determining spray drift and development of a model. Environ. Toxicol. Chem. 27, 2617–2626. https://doi.org/10.1897/08-109.1.

Wenneker, M., van de Zande, J., 2008. Spray drift reducing effects of natural windbreaks in orchard spraying. Asp. Appl. Biol. 84, 1–8.